Part III

Biophysical Aspects

Ecohydrology and Its Relation to Integrated Groundwater Management

12

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Abstract

In the twentieth century, groundwater characterization focused primarily on easily measured hydraulic metrics of water storage and flows. Twenty-first century concepts of groundwater *availability*, however, encompass other factors having societal value, such as ecological well-being. Effective ecohydrological science is a nexus of fundamental understanding derived from two scientific disciplines: (1) ecology, where scale, thresholds, feedbacks and tipping points for societal questions form the basis for the *ecologic* characterization, and (2) hydrology, where the characteristics, magnitude, and timing of water flows are characterized for a defined system of interest. In addition to ecohydrology itself, integrated groundwater management requires input from resource managers to understand which areas of the vast world of ecohydrology are important for decision making. Expectations of acceptable uncertainty, or even what ecohydrological outputs have utility, are often not well articulated within societal decision making frameworks, or within the science community itself. Similarly, "acceptable levels of impact" are difficult to define. Three examples

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are given to demonstrate the use of ecohydrological considerations for long-term sustainability of groundwater resources and their related ecosystem function. Such examples illustrate the importance of accommodating ecohydrogeological aspects into integrated groundwater management of the twenty-first century, regardless of society, climate, or setting.

12.1 Introduction

Groundwater resource characterization in the past was typically based on relatively easily estimated hydraulic metrics of water storage and flows within aquifers (see Chap. 3). This characterization occurred on smaller site scales to larger regional assessment, and employed well-established "classic" hydrogeological methods (Fig. 12.1). In the twenty-first century, however, such accounting approaches can miss a fundamental societal decision-making issue – there is no "unused" water in the environment (Hunt 2003). Because of mass balance, what is taken for a new use comes at the expense of an existing one. Recognizing the need to include this trade-off, groundwater resources are now evaluated in terms of water *availability*. In such a view, a more holistic view of the groundwater system is required, one that includes non-hydraulic factors such as ecological degradation. For example, although a shallow unconfined aquifer might have a saturated thickness of 100 s of meters, even small drawdowns can markedly change groundwater discharge to surface water features and associated ecological functions valued by society (Reilly et al. 2008). Thus, the system is characterized by large groundwater storage, but the storage actually *available* for use, as decided by non-hydraulic factors, is much less (Alley 2007).

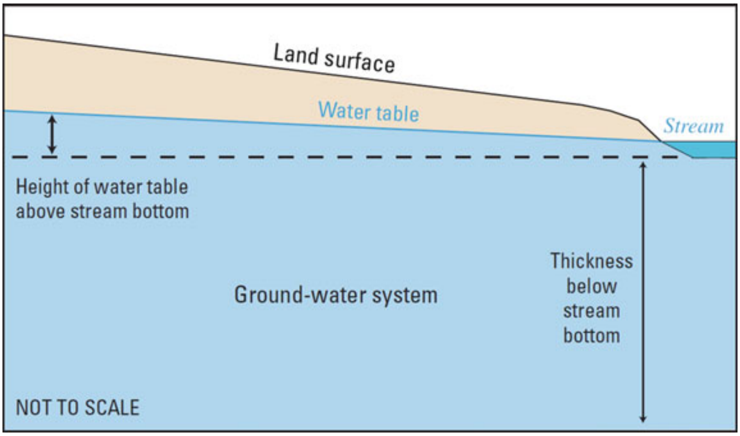

Fig. 12.1 An example of differences in water *stored* in an aquifer (large arrow on right) and the smaller amount of water *available* (small arrow on left) as determined by a societal desire to maintain surface water flow (Modified from Reilly et al. 2008)

Unconsolidated sediments in humid climates typify this issue. These deposits are commonly characterized by a high water-table elevation and high degree of connectedness/interaction between groundwater and surface waters, and in turn, associated ecosystems. Groundwater withdrawals from unconfined aquifers not only intercept groundwater that would discharge to surface waters and associated ecosystems, but can directly capture water from the stream under certain pumping conditions (Fig. 13, Alley et al. 1999). In either case, associated diversion of groundwater by drainage and well abstraction can be expected to stress local groundwater-dependent ecosystems (GDEs). In arid areas, the importance of groundwater is also seen, where the landscape singularities of higher moisture drive high levels of ecosystem production (Springer and Stevens 2009). Thus, a very small portion of the land surface can be responsible for the majority of the arid ecosystems' value. In these ways, groundwater is not only a resource to be exploited, but is also a hidden connector across the landscape (Hunt 2003). This connection transmits stress within the aquifer itself, and across and between surface waters (Winter et al. 1998) and many aquatic and terrestrial ecosystems. For GDEs, where the water table intersects or comes close to the land surface, the timing and magnitude of groundwater flow and related nutrient fluxes can be critical to ecosystem formation and persistence. Consider that precipitation is the dominant source of water in nearly all wetland systems in humid climates, yet the influence of the groundwater flow component can be sufficient (from an ecological perspective) to yield an entire new type of wetland community often valued by society, the fen (e.g., Amon et al. 2002). Influxes of groundwater to lakes, rivers, and wetlands can change whole-system physico–chemical properties (e.g., Anderson and Bowser 1986) including temperature and salinity, while also providing more subtle influences on microenvironments and ecological processes such as (e.g., Hurley et al. 1985). Infiltration of water from surface aquatic ecosystems also has a significant effect on aquifer ecology, especially on microbes and invertebrates (e.g., Hunt et al. 2006). Moreover, surface ecological processes such as evapotranspiration have long been recognized as potentially influencing hydrological responses (e.g., Meyboom 1964, 1966) and related hydrochemical function. Thus, the relation of groundwater hydrology to patterns and processes in ecology is a 'two-way street' where understanding the feedback of one to the other serves as a powerful lens through which to evaluate and explain the functioning of natural ecosystems (Hancock et al. 2009).

One difficulty for standard application of broad ecohydrological concepts to integrated groundwater management is that types of groundwater-ecology links can be wide-ranging – they can include the well-recognized relations found at the groundwater/surface-water interface such as water–plant interactions or groundwater–temperature–fish relations, but also less well-known topics such as microbial community characterization at the periphery of a contaminant plume. Thus, it is perhaps not surprising that standard ecohydrological procedures and metrics do not exist, and the significance and power of this ecohydrological tandem has not always been followed with effective interdisciplinary science. That is, the encompassing ecological, hydrological, and physico-chemical links between groundwater, surface waters and associated ecosystems are seldom fully understood, even though true characterization and optimal management may require such an encompassing multidisciplinary view. Shortcomings in our ability to perform true characterization notwithstanding, overarching concepts of application to integrated groundwater management can be developed, and much can be learned from successful (and unsuccessful) attempts at ecohydrology.

One way to characterize the overarching interplay between ecology and hydrology is this: consideration of ecohydrological issues enhances understanding amongst biologists, as hydrogeology provides the abiotic "box", within which ecological processes play out. Biologists and ecologists articulate defining characteristics of groundwater flows required for their societally relevant target – insight that requires the skills of hydrogeologists to attain. Hydrogeologists, in turn, must understand how and why groundwater influences ecological processes so that their expertise is effectively brought to bear on the ecological question (Hunt and Wilcox 2003a, b). Moreover, hydrogeologists have to recognize that the ecological system can influence the groundwater system most notably by evapotranspiration from shallow groundwater (Batelaan et al. 2003). Ecological factors help define important spatial and temporal scales, which in many cases are smaller than classical hydrogeologic characterization. In addition, ecological factors facilitate identification of qualitative levels of certainty needed in abiotic characterizations. Learning about ecological *thresholds* and *tipping points* for the societal question at hand helps define the work needed, and ensure it is tackled efficiently. An ecological threshold can be described as a system condition whereby a small change in external conditions causes a rapid change in an ecosystem, and passing the ecological threshold leads to rapid change of ecosystem health. An ecological tipping point is where the change moves from one stable state to another stable state, often irreversibly. To understand how a threshold can influence decision making, consider the selection of a pipe sized to convey a well's pumpage that is somewhat uncertain (Hancock et al. 2009): pipe sizes come in a set range of diameters so estimated pumpage is evaluated with the pipe-size thresholds in mind. If one is relatively certain that a pipe diameter (threshold) will not convey the estimated pumpage, then a larger diameter of pipe is chosen. Knowledge of pipe-size thresholds simplifies and directs the question into a form much different than trying to estimate the exact rate of well pumpage itself. In a similar ecohydrological context, the ecological threshold of a stream drying up is a very different abiotic forecast than estimating various degrees of low flow in a perennial stream. Therefore, the ecological threshold can simplify and direct the types of hydrogeological investigation brought to bear to characterize the system appropriately.

Identification of which thresholds and tipping points are societally important is often provided by the resource managers, and thus can be considered an important link for effective integrated groundwater management. A societal context for science has become increasingly important (e.g., Boland 2010; Guillaume et al. 2012); resource managers are better acquainted with competing needs and rank of societally valued ecosystem services. Thus, they are critical for including in the discussion of tradeoffs of one *versus* another, and ranking which areas of ecohydrology are societally relevant for decision making. Their input elucidates connections between groundwater and terrestrial/subterranean ecosystems that facilitate holistic management of natural systems, and helps create a complete listing of the threats and mitigation opportunities. Such input moves discussions of water availability to long-term *sustainability* of the resource and its ecosystem goods and services. Such a multi-discipline approach is needed to effect true integrated groundwater management.

In this chapter a historical background and examples of ecohydrology and integrated groundwater management are provided with these considerations in mind. Because the range of potential societally relevant ecological endpoints is vast, we focus on transferable elements contained within the examples rather than problem-specific insight. The chapter concludes with discussion of concepts and approaches for including ecohydrological considerations into integrated groundwater management. Using the dimensions of integrated groundwater management outlined in Chap. 1, ecohydrology can be seen as integration of multiple disciplines assessing natural and human systems across multiple scales of space and time. This integration, in turn, gives an encompassing foundation for discussion involving stakeholders, resource managers, and decision-makers. It should be noted that the topic of groundwater dependent ecosystems is sufficiently large and important for integrated groundwater management that it warrants its own chapter (Chap. 13). Therefore, these important systems are discussed only cursorily here.

12.2 Background of Ecohydrology and Water Management

In the last 10 years ecohydrology has been developed as a new scientific discipline. Recently its importance has been stressed in relation to hydro(geo)logy and ecology but also a wider range of ideas within the broad field of "ecohydrology". Elements of the history of ecohydrology are described here, which provide a foundation for the role of ecohydrology in groundwater management.

Several definitions of "ecohydrology" have been published:

- Wassen and Grootjans (1996): 'An application driven discipline aiming at a better understanding of hydrological factors determining the natural development of wet ecosystems, especially in regard of their functional value for natural protection and restoration'.
- Baird and Wilby (1999): 'Eco-hydrology is the study of plant-water interactions and the hydrological processes related to plant growth'.
- Eamus et al. (2006): 'Ecohydrology is the study of how the movement and storage of water in the environment and the structure and function of vegetation are linked in a reciprocal exchange.'
- Rodriguez-Iturbe (2000): 'Eco-hydrology seeks to describe the hydrological mechanisms that underlie ecological pattern and processes'.

- Nuttle (2002): 'Eco-hydrology is . . . concerned with the effects of hydrological processes on the distribution, structure and function of ecosystems, and on the effect of biological processes on the elements of the water cycle'.
- Hunt and Wilcox (2003a): 'ecohydrology (is) defined . . . as tightly coupled research in which both (ecology and hydrology) disciplines are equally involved in the formulation of the research objective, design of the work plan, and on-going interpretation.'

The range of definitions clearly shows an imprint of the background from which different authors approach the field of ecohydrology, ranging from wetlands (nature protection), plant-water interaction, and, more recently, emphasis on bi-directional understanding provided by integrated application of hydrology, micrometeorology, and ecology.

Since 2000, ecohydrology has become popular in hydrological literature, including both dryland hydrology such as soil moisture-limited evapotranspiration processes (Rodriguez-Iturbe 2000; Eagleson 2002; Rodriguez-Iturbe and Porporato 2004), effects of streamflow and temperature on temperate climate biotic communities (e.g., Boulton and Hancock 2006; Hunt et al. 2006; Olson and Young 2009), and even using viruses as tracers of groundwater flow (Hunt et al. 2014). This recent interest in ecohydrology notwithstanding, much is to be gained by consideration of pre-2000 ecohydrological research roots, some of which have clear groundwater, and groundwater management, origins.

Early humans undoubtedly had some ecohydrological consciousness, as the recognition of certain plant species warned him of dangerous places where he could drown, or offered opportunities to find food. One of the earliest transcripts reporting on the topic comes from the Bible. Ross (2007) interprets and translates the Hebrew bible text of Isaiah 44 in modern language as: 'I will pour out My spirit as suddenly and overwhelmingly as a rainstorm in the desert. After such a storm, the willow does not fade like grass, but is kept green for many years by groundwater that recharges in the storm'. Obviously, this expresses some form of early 'ecohydrological' observations relating rainfall-recharge-groundwater with plant species occurrence. Vitruvius (15 BC) (1913), roman architect and engineer in the first century, wrote this concerning exploration of drinking water: 'One of the indications where groundwater can be found is the occurrence of small rushes, willows, alder, vitex, reeds and ivy'. Moreover he remarks: 'one must not rely on these plants if they occur in marshes, which receive and collect rain water'. Hence, he was well aware of the relativity of the plants as indicators for good quality groundwater, differences between sources of water, and the usefulness of ecological indicators for groundwater-drinking water management.

In tenth century AD, Mohammed ibn al-Hasan al-Hasib al-Karaji included a more holistic consideration of the subsurface into ecohydrology. Karaji was a mathematician and engineer who mainly lived and worked in Baghdad. In an effort to support the water resources exploration of his native Persia during the later stages of his career, he wrote the book 'The Extraction of Hidden Waters to the Surface', which is regarded the oldest textbook in hydrology/groundwater science (Nadji and

Voigt 1972; Pazwash and Mavrigian 1980). In the book, Karaji includes techniques for the exploration of groundwater such as wells and qanats – techniques still used today in many parts of the Middle East and Asia. He also examines how plants indicate the presence of groundwater by studying the roots of plants and how they grow towards the water table, and includes a report of a well digger who found roots at a depth of 50 m (Nadji and Voigt 1972; Pazwash and Mavrigian 1980). From this treatise, it is clear that Karaji had a surprising good understanding of groundwater and hydrological processes, and used this understanding to further develop ecohydrological relationships with vegetation.

In the mid-nineteenth century, the famous work of engineer Henri Darcy revolutionized this early understanding of groundwater flow. Often overlooked in the pioneering work of Darcy (1856), however, is the fact that it contains a description for the search for drinking water by spring seeker Father Paramelle, in addition to the much better-known column tests. Darcy relates how Father Paramelle infers the probable presence of water, and even the approximate depth of the water below the ground surface, from the nature and strength of the plants. Paramelle (1859) documents his methods in detail, which are notable for using a multidisciplinary approach that includes careful observation and evaluation of geology, mineralogy, topography and vegetation. His methods provided water for many communities in France, where he identified more than 10,000 springs.

Later in the nineteenth century, botanist A.F.W. Schimper focused on the detailed knowledge of plants and their specific habitats, and illustrated an important distinction between wet, hygrophyte and dry, xerophyte plant species. The difference lies in the plant physiology: if a soil contains too much salt, the plants cannot absorb the water and hence it is physiologically dry. All soils which are physically dry are also physiologically dry; and hence only the physiological dryness or wetness of soils need be considered in ecology of plant communities near the ends of this gradient (Schimper 1898). O.E. Meinzer, the father of modern groundwater hydrology, was the first to define the term phreatophyte as a plant that habitually obtains its water supply from groundwater (Meinzer 1923). In 1927, he wrote an entire book about these phreatophytes (Meinzer 1927). In it, he describes the principal phreatophytic species, like common salt grass *(Distilchlis* spicata) and their occurrence in the arid and semi-arid regions of the US. With this understanding, Meinzer and other groundwater hydrologists could then use plants as indicators for locations of groundwater resources.

After the first half of the twentieth century, the use of phreatophytes in groundwater studies became less prominent in the hydrogeological literature; however, ecologists continued the study of plant habitat requirements (Londo 1988; Ellenberg et al. 1992). Ecologists interested in plant community composition, development, and species relations ("phytosociologists") started to research the relationship between vegetation types and groundwater dynamics in the 1950s. Ellenberg (1948, 1950, 1952, 1953, 1974) and Tüxen (1954) systematically studied the relation between groundwater level and the occurrence of vegetation types. More recently, interest in phreatophytes again became a prominent topic of study following the interest and formal need for protection (European Union 2000, 2006) of groundwater dependent ecosystems (Batelaan et al. 2003; Witte and von Asmuth 2003; Loheide et al. 2005).

The first publication in which the word 'ecohydrology' is mentioned is from the Dutch author van Wirdum (1982), and came about through a groundwater management concern. In van Wirdum's annual report of the activities of the Dutch National Institute for Nature Research, one sees a growing recognition for ecological values of wetlands (Grootjans et al. 1988; Wassen et al. 1990; Mitsch and Gosselink 1993). This recognition was driven by an observed deterioration of high ecological functions of wetlands due to poor water management. For example, desiccation resulting from groundwater abstraction and agricultural drainage, along with water pollution (Schot et al. 1988) were identified as important factors reducing biodiversity. Hence, even with this early use of the word 'ecohydrology' it was understood that groundwater management could significantly influence ecological values.

12.3 Examples of Ecohydrology and Water Management

Groundwater has well-recognized ecological functions including: (1) sustaining stream base flow and moderating water-level fluctuations of groundwater-fed lakes and wetlands, (2) providing stable-temperature habitats, (3) supplying nutrients and inorganic ions, and (4) providing moisture for riparian and other groundwater-dependent vegetation (Hayashi and Rosenberry 2002). The importance of these functions is being incorporated in water management policies of European countries through the Water Framework and Groundwater Directives (European Union 2000, 2006), and has been gaining recognition in other parts of the world over the past decade or so. The following three examples explore how the considerations of the interaction of the ecosystem with the groundwater system influenced management of the resource.

12.3.1 Temperate Climate: United Kingdom

The European Water Framework and its progeny Groundwater Directive require assessment of the status of groundwater bodies with respect to various criteria including the condition of a groundwater-dependent terrestrial ecosystem. Using wetlands as an example of a groundwater-dependent ecosystem, Whiteman et al. (2010) describe a screening tool to assess wetland condition by examining three factors: (1) condition of source groundwater (rate of abstraction, concentration of contaminants, etc.), (2) connectivity between groundwater and the wetland, and (3) ecological response of the wetland to changes in hydrological condition. By assigning scores to the three criteria at 1,368 test sites in England and Wales, they identified 63 wetlands as having high risk from abstraction pressures and 117 from contamination pressures. Once a potentially high-risk site is identified, site-specific investigation is initiated to assess the actual condition of groundwater and, if poor condition is confirmed, potential mitigation measures are explored.

For example, Hurcott and Podmore Pools is a series of pools and marginal wetlands within a large alder wetland/woodland in Worcestershire (Whiteman et al. 2010). The main sources of water for the pools are surface water inflows from the upstream catchment and groundwater discharge from a major sandstone aquifer, which is also an important public water supply. Unsustainable groundwater abstraction from the aquifer caused a wide-scale drawdown of water levels in the aquifer (poor condition of source water), which significantly reduced stream inflows to the site and eliminated direct groundwater discharge to the site (poor connectivity), and which in turn resulted in a measurable change in vegetation community composition (ecological response). Detailed site assessment suggested that summer maximum water-table depths should be less than 0.45 m to support the ecosystem; however, water-table fluctuations up to 0.7 m were observed. Based on these observations and numerical groundwater modeling results, a Water Level Management Plan (Whiteman et al. 2010) was proposed and implemented to raise the water table and to potentially change the groundwater abstraction regime.

12.3.2 Semi-Arid Climate: Kansas, United States

The State of Kansas in the USA has a long history of integrated groundwater management, which provides a useful case study to demonstrate the paradigm shift in groundwater management. The following summary of the Kansas case study has been largely drawn from a major body of work by Marios Sophocleous at the Kansas Geological Survey. Irrigation is the largest user of water in Kansas, accounting for 80–85 % of total water use (KWO 2009), most of which comes from groundwater extracted from the High Plains aquifer. Groundwater abstraction rapidly increased after the enactment of the Kansas Water Appropriation Act in 1945, which permitted water rights to users for "beneficial use" (Sophocleous 2011). By the late 1960s, too many water rights had been permitted, enabling over-development of the High Plains aquifer resulting in the mining of groundwater resources (Sophocleous 1998). To prevent further mining of groundwater, five Groundwater Management Districts (GMDs) were established, covering most of the extent of the High Plains aquifer, and a "safe-yield" management policy was adopted in the GMDs (Sophocleous 2000).

The aim of this management policy was to balance groundwater withdrawals with aquifer recharge by limiting the total water abstraction in a 3.2-km circle around any proposed new abstraction to be less than the long-term average annual recharge (Sophocleous 2000). This policy had an effect of slowing the rate of water-table declines in the aquifer, but the policy did not stop the decline. More importantly, the safe-yield concept was known to be problematic in practice (e.g., Thomas and Leopold 1964) as it gives no consideration to maintaining naturally occurring groundwater discharge that sustains the perennial flow of streams (Sophocleous 1997). As a result, stream flows and associated riparian and aquatic ecosystems in western and central Kansas steadily declined and the related ecosystem deteriorated (Sophocleous 1998).

Recognizing that streams and aquifers are closely linked and have to be understood and managed together, in the early 1990s some of the GMD's moved toward *conjunctive* stream-aquifer management by including baseflow in the evaluation (Sophocleous 2000). In other words, baseflow is considered a societal value that it has been given a water right on its own. This shifts the focus from the problematic *aquifer* safe-yield paradigm to a more holistic sustainable *system* water management paradigm. It was hoped that the new measure, together with the legal establishment of minimum-desirable streamflow standard in 1984, would provide needed protection to the riverine-riparian ecosystem (Sophocleous 2011). As a result of GMD actions, pumping rates of groundwater in Kansas leveled off after decades of increases. However, the aquifer had already been mined to a significant reduction of saturated thickness and many streams had deteriorated due to earlier over-development (Sophocleous 1998). The long-term goal of the GMDs is to reduce the rate of water use in order to prolong the life of the aquifer and maintain the remaining groundwater-dependent terrestrial ecosystem. Towards this goal, Intensive Groundwater Use Control Areas (IGUCAs) were established in locations where unfavorable conditions existed, including situations where groundwater use was adversely depleting streamflow and adversely affecting ecology (Sophocleous 2011). Such a tiered designation is a powerful tool that allows the use of a variety of measures, including the reduction in existing water rights, to solve groundwater and ecological issues.

In addition to the revised safe-yield policy explicitly considering baseflow and the use of IGUCAs, Kansas has been using innovative measures to enhance the riverine-riparian ecosystems. For example, private, not-for-profit water-bank systems are used to provide open-market approach for temporarily moving water rights from inactive users to active users (Stover et al. 2011). The Conservation Reserve Enhancement Program is used to give economic incentive to owners of irrigated land to retire lands located in sensitive areas, for example along river corridors of drying streams (Leatherman et al. 2006). In order to enhance integrated water management of groundwater-dependent ecosystems, Sophocleous (2007, 2011) suggests that: (1) the definition of "beneficial" water use must be expanded from traditional irrigation and other consumptive uses to include water conservation and instream flow needs; (2) domestic and other wells that are currently exempted must be included with regulated uses; and (3) increased flexibility of regulatory requirements regarding transferring water rights is needed.

12.3.3 Arid Climate: Australia and United States

The above examples demonstrate groundwater management efforts to support riverine-riparian ecosystems. In some water-scarce regions, however, riparian trees were deemed harmful to stream ecosystems because they take up and transpire groundwater that would otherwise be available to sustain baseflow. Doody et al. (2011) present a review of case studies from the western USA and south-eastern Australia, where removal of non-native riparian vegetation has been attempted to reduce transpiration diversion and enhance stream flow. In the western USA, non-native phreatophyte, saltcedar (*Tamarix spp.*) had spread along many river bottoms by the 1950s and became a primary target of water "salvage" projects (Robinson 1965). Contrary to the original perception that saltcedar had a higher water use compared with native species, many years of studies including large-scale tree removal experiments have shown that the reduction in evapotranspiration by the removal of saltcedar had no measureable effect on streamflow. This surprising finding was due to similar transpiration rates between the non-native saltcedar and the vegetation community that was established after the saltcedar was removed. In this case, unexpected ecological aspects confounded the expected hydrologic response. Other ecohydrological work showed similar results: no large-scale removal experiments in arid settings have shown the expected return of increased stream flow.

In the Murray-Darling Basin in south-eastern Australia, colonization of the non-native phreatophyte, willow (*Salix spp.*) has also been associated with a number of undesirable impacts on stream ecosystems, including increased water uptake and transpiration, and subsequent reduction in streamflow. Similar to the United States saltcedar, site-scale studies have shown that willows growing in the riparian zone have evapotranspiration rates similar to native *Eucalyptus spp.*, suggesting that removing willow from stream banks will have little effects on net stream flow (Doody et al. 2011). However, unlike saltcedar, willow growth has other hydrologic effects beyond capture and transpiration of groundwater that would discharge to streams. That is, it also grows *within* wet stream channels and reduces flow velocity. The reduction in velocity facilitates water capture, and because the willow is rarely water limited, they can transpire at a higher rate than open-water evaporation (Doody and Benyon 2011). Because the native *Eucalyptus* more commonly grows on river banks and not in the channel, the removal of willow from within stream channel is expected to result in significant water salvage. These examples indicate the importance of understanding eco-hydrological processes specific to the problem – in this case water uptake by trees – to design effective methods of integrated groundwater management.

12.4 Incorporating Ecohydrology into Integrated Groundwater Management

Taken as a whole, concepts and approaches discussed above lead to salient insight into how ecohydrological considerations can be integrated with groundwater management.

1. Groundwater *availability* constraints in highly connected groundwater and surface water systems are a function of both ecosystem degradation *and* water-use

needs. Even though the latter is often an initial primary driver, aspects of the former often become key drivers for decisions of allowable water use. Therefore, *sustainability* of the groundwater resource can be expected to be tied to ecohydrological drivers.

2. Hydrologic measurements (e.g., streamflow statistics or water-table depth in a wetland) allow decision makers to obtain quick snapshots of the system of interest. These "sentinel metrics", however, are often an indirect measure of what is considered valuable. Therefore, these sentinels need to be formally recognized by stakeholders as surrogates for societally-relevant system qualities. The identification of a set of surrogate sentinel metrics is critical for integrated groundwater management because full system characterization after each management change is not feasible. Moreover, the real-time insight of properly identified sentinel metrics can move an adaptive management plan from simple monitoring to proactive actions that can mitigate ecosystem degradation.
3. Many integrated groundwater management questions are complex – both in ways systems interact as well as feedback mechanisms that mitigate or exacerbate the effect of potential change. Such questions may require hydrologic or ecologic characterizations that are more holistic and comprehensive than sentinel metrics. The goal of this higher level of ecohydrological work is development of a quantitative framework for how much degradation can be expected for differing levels of groundwater withdrawals (or diversions). This allows quantification of the trade-offs inherent to ecohydrology, which in turn can inform cost-benefit analyses conducted by stakeholders. Characteristic functions of ecosystem response, such as response curves (e.g., GCAC 2007; Chap. 6), thresholds, and tipping points, for species of interest give language and help visualize tradeoffs between water use and ecosystem degradation – evaluations inherent to integrated groundwater management.
4. There is a need to translate each science and resource manager concern into terms and metrics that are understandable to all involved. Ecologists may resist having their science being held to the precision that hydrogeologists routinely report, yet are comfortable focusing on thresholds and tipping points for their ecosystem. Successful integrated groundwater management will, in large measure, be a reflection of how well the interaction between ecology and hydrology aspects is articulated.
5. Similar to an adaptive management framework, integrated groundwater management must recognize that many of the underlying feedback loops and system complexity will never be fully understood, especially given the relatively short timeframe of most decision-making. Yet, just as with the adaptive management approach to handling confounding uncertainty, the integrated groundwater management framework can form the crucible of hypothesis testing, where it distills all possible ecohydrological research topics to a subset that can be prioritized. In this way, integrated groundwater management provides a relevance that may be missing in simple academic ecohydrological endeavours. An effective integrated groundwater management plan is expected to include aspects of applied research that focuses on spatial and temporal scales relevant to both the hydrogeological

and the ecological process being studied. This is important to note since historically, hydrogeological studies often are performed on the aquifer or site scale, thus using approaches and generating data too broad for understanding many ecological processes on a site or smaller scale. Moreover, both hydrogeological and ecological foci may have not been optimally tuned for the resource management question of primary interest.

12.5 Summary

In summary, the demands of twenty-first century integrated groundwater management might be considered to precede the maturation of ecohydrological science, a view that might be concluded from the lack of dominant textbooks published or widely accepted common guidelines. However, we believe there are many necessary and common elements in current science methods that have direct application to today's integrated groundwater management. Moreover, formally including societal drivers as the basis for ecohydrological action provides an important foundation for effective ecohydrology in the twenty-first century. Such a focus can only help move the societally relevant and necessary science of ecohydrology into effective integrated management.

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Groundwater Dependent Ecosystems: Classification, Identification Techniques and Threats

13

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Abstract

This chapter begins by briefly discussing the three major classes of groundwater dependent ecosystems (GDEs), namely: (I) GDEs that reside within groundwater (e.g. karsts; stygofauna); (II) GDEs requiring the surface expression of groundwater (e.g. springs; wetlands); and (III) GDEs dependent upon sub-surface availability of groundwater within the rooting depth of vegetation (e.g. woodlands; riparian forests). We then discuss a range of techniques available for identifying the location of GDEs in a landscape, with a primary focus of class III GDEs and a secondary focus of class II GDEs. These techniques include inferential methodologies, using hydrological, geochemical and geomorphological indicators, biotic assemblages, historical documentation, and remote sensing methodologies. Techniques available to quantify groundwater use by GDEs are briefly described, including application of simple modelling tools, remote sensing methods and complex modelling applications. This chapter also outlines the contemporary threats to the persistence of GDEs across the world. This involves a description of the "natural" hydrological attributes relevant to GDEs and the

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processes that lead to disturbances to natural hydrological attributes as a result of human activities (e.g. groundwater extraction). Two cases studies, (1) Class III: terrestrial vegetation and (2) Class II: springs, are discussed in relation to these issues.

13.1 Introduction

In order to sustainably manage groundwater in a truly integrated manner consideration needs to be given to the interaction of groundwater with ecology. Groundwater interacts with multiple classes of biome, including stygofauna of aquifers, rivers relying on base flow (the discharge of groundwater into rivers) and terrestrial ecosystems. Management plans that do not include such consideration are likely to negatively impact these groundwater dependent ecosystems.

In this chapter, we focus on the links between ecology and groundwater availability, rather than on groundwater resources and human demand. This is because we feel that environmental allocations of groundwater have generally received less attention than allocations to human demands and because we identify four important knowledge gaps to the sustainable management of environmental allocations of groundwater. These are:

1. How do we know where a GDE is in the landscape? If we do not know where they are, we cannot manage them and allocate groundwater resources appropriately.
2. How much groundwater is used by a GDE? If we do not know how much groundwater is used, we cannot allocate an appropriate quantity of the resource.
3. What are the threats to GDEs? Only by understanding the threats to GDEs can we ensure their sustainable management.
4. What are the likely responses of GDEs to over extraction of groundwater? Without knowing what to measure, we cannot regulate groundwater extraction in ways that do not negatively impact on GDEs.

13.2 Classes of GDEs and Relevant Groundwater Attributes

13.2.1 GDE Classification

Hatton and Evans (1998) were perhaps the first to attempt to categorise GDEs systematically. They recognised five classes of ecosystem dependency on groundwater:

1. Ecosystems entirely dependent on groundwater; or obligate GDEs. In these communities only small changes in groundwater availability or quality result

in the total loss of the current ecosystem structure and function. Examples of entirely dependent ecosystems include the mound spring systems of the Great Artesian Basin of eastern Australia, karstic groundwater ecosystems of Western Australia and riparian vegetation along streams in central Australia.

2. Ecosystems highly dependent on groundwater. In these communities small to moderate changes in groundwater availability result in significant changes in ecosystem structure and function. Examples of highly dependent ecosystems in Australia include: *Melaleuca* swamp forests and woodlands of tropical northern Australia, base flow dependent ecosystems of temperate Australia and the damp lands of the Swan Coastal Plain.
3. Ecosystems with proportional dependence on groundwater. Such ecosystems do not exhibit the threshold-type responses of (1) and (2) above. As groundwater availability or quality changes, there is a proportional response in ecosystem structure and function and distribution. Examples include base flow and permanent lake ecosystems.
4. Ecosystems that are opportunistic users of groundwater. In these ecosystems groundwater has a significant role in their water balance occasionally and reliance is not obligate (so-called facultative dependency). Examples of opportunistic ecosystems include swamp forests of coastal floodplains along the fringe of the south-east uplands and Jarrah forests and *Banksia* woodlands of Western Australia.
5. Ecosystems that appear to be groundwater dependent, but are in fact entirely rain fed or dependent only on surface water flows. Examples of this type include seasonal floodplain lakes on small creeks in northern Australia and terminal drainage basin lakes in the Central Lowlands.

There are two major problems with this classification system. First, the determination of the degree of dependency is difficult and requires many years of study of a site. Establishing that an ecosystem is only an opportunistic user of groundwater may require a decade of waiting before a drought occurs and groundwater dependency becomes expressed. Second, establishing the presence or absence of a threshold response is extremely difficult and time consuming. Consequently, a simplified classification system was proposed by Eamus et al. (2006):

(I) Aquifer and cave ecosystems where stygofauna reside. This class also includes the hyporheic zones of rivers and floodplains.
(II) Ecosystems reliant on surface expression of groundwater. This includes base flow rivers, streams and wetlands, springs and estuarine seagrasses.
(III) Ecosystems reliant on sub-surface presence of groundwater within the rooting depth of the ecosystem (usually via the capillary fringe).

Application of this simple classification scheme assists managers in identifying the correct techniques for assessing GDE structure, function and management regime (Eamus et al. 2006). This classification scheme was recently adopted in the Australian National Atlas of Groundwater Dependent Ecosystems.

13.2.2 Classification of Springs Ecosystems as GDEs

Springs occur in geomorphic settings that are far more complex than those of most wetlands, emerging from hill slopes, cliff faces, and beneath other bodies of water. Adding to their complex emergence environment, springs often support a wide array of microhabitats not observed in wetlands. The "sphere" into which the aquifer discharges was initially described by Meinzer (1923), and then simplified by Hynes (1970) into three classes: rheocrene (channel emergence), limnocrene (pool emergence), and helocrene (wet meadow emergence).

Springer et al. (2008) and Springer and Stevens (2009) reviewed literature and expanded this historical scheme to include 12 spheres of discharge of terrestrial springs, including: (1) springs that emerge in caves, (2) exposure springs, (3) artesian fountains, (4) geysers, (5) gushets, (6) contact hanging gardens, (7) helocrene wet meadows, (8) hill slope springs, (9) hypocrene buried springs, (10) limnocrene surficial lentic pools, (11) mineralized mounds, and (12) rheocrene lotic channel floors. This classification provides a more precise lexicon with which to describe groundwater emergence function in relation to ecosystem landform configuration and distribution.

Geomorphological variation among the 12 terrestrial springs types of Springer and Stevens (2009) leads to predictable variation in spring's vegetation, habitat structure, plant and faunal diversity, and ecosystem structure and function (Griffiths et al. 2008). For example, helocrene springs are typically dominated by wetland graminoid and shrub species, with little canopy cover by trees. Many hill slope springs typically occupy a position on the landscape where groundwater discharge has created a shallow concave depression due to low discharge rates winnowing away fine-grained sediments or groundwater sapping to create spring dependent headwater theatres for channels (Laity and Malin 1985; Meinzer 1923).

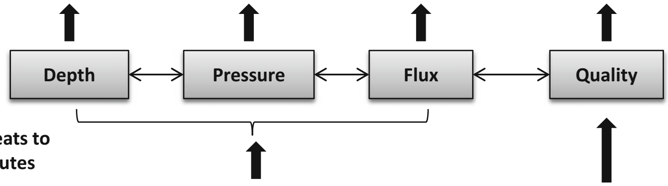
13.2.3 Relevant Groundwater Attributes

The persistence of GDEs relies on suitable groundwater attributes. Identifying these attributes is essential as this can help establish groundwater management targets and monitoring strategies (Kreamer et al. 2014). In general, the following groundwater attributes are important for GDEs (Clifton and Evans 2001):

1. Depth-to-groundwater, for unconfined aquifers;
2. Groundwater pressure – hydraulic head and its expression in groundwater discharge, for confined aquifers;
3. Groundwater flux – flow rate and volume of groundwater supply; flow direction;
4. Groundwater quality – including groundwater salinity, acidity and the concentrations of nutrients and pollutants.

Importance of groundwater attributes to GDEs

Class I GDEs (e.g. woodlands)	• Accessible water at root zones; • Prevent water-logging.		• Sustain water uptake rate.	• Maintain suitable chemical composition in water supply.
Class II GDEs (e.g. wetlands, streams)	• Provide wetness or water-logged environment; • Prevent activation of acid sulphate soil; • Maintain hydraulic gradient for groundwater discharge.	• Sustain groundwater discharge to springs.	• Sustain above ground wetness (wetlands); • Sustain base flow; • Prevent salt water intrusion (estuarine/coastal environment).	• Maintain suitable chemical composition in water supply and living environment.
Class III GDEs (e.g. cave systems)	• Provide living habitat; • Maintain groundwater stratification.		• Supply organic matter and oxygen.	• Maintain suitable chemical composition in living environment.



Anthropogenic threats to groundwater attributes

Agricultural practices	• Reduced groundwater level/pressure due to excessive groundwater extraction to support agricultural development; • Reduced groundwater recharge due to surface water pumping for irrigation; • Water-logging due to vegetation clearing and poorly managed irrigation.	• Groundwater contamination from fertilisers, pesticides and other agricultural chemicals. • Soil and water salinisation due to vegetation clearing and excessive irrigation.
Urban and industrial development	• Reduced groundwater level/pressure due to excessive groundwater extraction to support urban and industrial development.	• Ground water contamination from urban facilities, landfills, fertilisers and pesticides (e.g. for gardens and parks), stormwater/sewage disposal, and other industrial chemicals.
Mining activities	• Reduced level, pressure and flux due to mine dewatering; • Reduced level due to channel incision (e.g. gravel mining)	• Change in groundwater stratification due to dewatering; • Groundwater contamination from tailings dams; • Groundwater contamination through leaching of acidic or toxic crushed rock storage sites; • Groundwater contamination after mine closure, due to water table rise and mine flooding.
Plantation forestry	• Reduced groundwater recharge and surface flow; • Increased groundwater discharge.	

Fig. 13.1 Importance of groundwater regime (depth-to-groundwater and groundwater pressure and flux) and quality on different classes of GDEs and the anthropogenic threats

Importance of these attributes to GDEs is summarised in Fig. 13.1. Depth-to-groundwater (from the land surface) is perhaps one of the most important groundwater attributes for GDEs (Eamus et al. 2006). This is particularly the case for terrestrial ecosystems that rely on sub-surface provision of groundwater. Depth-to-groundwater, with particular reference to the distance between the capillary fringe above the water table and plant roots, directly determines groundwater availability to vegetation. An increased depth-to-groundwater may lead to reduced plant growth, mortality and change in species compositions (Shafroth et al. 2000). Lowering a water table can also lead to loss of habitat for cave and aquifer ecosystems (Boulton et al. 2003; Heitmuller and Reece 2007). On the other hand, a rising water table may disadvantage those species vulnerable to water-logging and lead to succession to different plant communities (Naumburg et al. 2005). Changes in water table depth, coupled with other environmental factors, can also result in groundwater contamination. For example, lowering a water table beneath acid sulphate soils leads to oxidation of pyrite and subsequent acidification of the shallow aquifer (Ritsema et al. 1992; Nath et al. 2013).

Groundwater flux is important for Classes II and III GDEs because it sustains water uptake by vegetation (Shafroth et al. 2000). Reduced groundwater pressure and flux cause reduced groundwater discharge and subsequently reduced surface water availability to wetlands and GDEs that depend on base flow and springs (Zektser et al. 2005). In estuary or coastal areas reduced groundwater flux leads to seawater intrusion and contamination of coastal freshwater aquifers (Jayasekera et al. 2011; Lambrakis 1998), thereby reducing groundwater quality. For cave and aquifer ecosystems, appropriate groundwater flux is important to maintain a supply of organic matter and oxygen (Hancock et al. 2005) to stygofauna contained within these systems. Groundwater quality is critical for all types of GDEs to maintain suitable chemical composition in water supply and/or living environment. In some areas, groundwater is hydrochemically stratified. Disturbing the stratification may cause the chemical composition to be unsuitable for the associated aquifer ecosystems.

Depth-to-groundwater and groundwater pressure and flux naturally fluctuate. In unconfined aquifers, short-term fluctuations naturally occur in response to time-varying uptake of water by vegetation; whereas longer term fluctuations often reflect time-varying groundwater recharge as a result of wet and dry season cycles. GDEs that are developed at naturally highly fluctuating areas (e.g. areas with strong climatic seasonality) generally have adapted to the fluctuations of groundwater regime and hence can be more resilient to change in groundwater regime than those developed from areas with more constant regime. For example, in the Howard River catchment of the Northern Territory of Australia, natural intra-annual variation in groundwater depth is approximately 8 m (Cook et al. 1998). This large variation (arising through a combination of wet and dry season variation in rainfall, lateral sub-surface flow of groundwater to the Howard River and evapotranspirational discharge) is accommodated through changes in landscape leaf area index (LAI) and root depth.

These groundwater attributes can be altered due to human activities. The contemporary threats to the persistence of GDEs, including the processes that lead to disturbances to natural hydrological attributes as a result of human activities (e.g. groundwater abstraction), are described in Sect. 13.5.

13.3 Identifying GDEs

Identifying the location of GDEs and assessing their dependency on groundwater is the vital first step to managing them. However, identifying their location across a landscape is often difficult, time-consuming and hence expensive and always requires a high level of technical expertise. In this section, a range of techniques that can be used to assist in this are discussed.

13.3.1 Inferential Methods to Determine GDEs

Early assessments of groundwater dependency frequently relied on inference (Clifton and Evans 2001; Eamus et al. 2006). Thus, answers in the affirmative to one or more of the following can be taken as supporting the hypothesis that at least some species in an ecosystem are using groundwater.

1. Does a stream/river flow all year, despite long periods of low or zero rainfall (and thus zero surface flows)?
2. For estuarine systems, do salinity levels fall below that of seawater in the absence of surface water inputs?
3. Does the total flux in a river increase downstream in the absence of inflow from a tributary or surface flow?
4. Are water levels in a wetland maintained during extended dry periods?
5. Is groundwater discharged to the surface for significant periods of time each year? If such a resource is present, evolution will ensure that some species will be using it.
6. Is the vegetation associated with the surface discharge of groundwater different (in terms of species composition, phenological pattern, leaf area index or vegetation structure) from vegetation close-by but which is not accessing this groundwater?
7. Is the annual rate of transpiration by vegetation at a suspected GDE significantly larger than annual rainfall at the site and the site is not a run-on site?
8. Are plant water relations (especially pre-dawn and mid-day water potentials and transpiration rates) indicative of less water stress (water potentials closer to zero; transpiration rate larger) than vegetation located nearby but not accessing the groundwater discharged at the surface? The best time to assess this is during rain-less periods.
9. Does the water balance of a site indicate that the sum of water-use plus interception loss plus run-off plus deep drainage is significantly larger that annual rainfall plus run-on?
10. Is occasional (or habitual) groundwater release at the surface associated with key developmental stages of the vegetation (such as flowering, germination, seedling establishment)?

11. Does groundwater and hydrological modelling suggest that groundwater is either discharging to the surface or located within the likely rooting depth of the vegetation?
12. Is groundwater or the capillary fringe above the water table present within the rooting depth of any of the vegetation?
13. Does a proportion of the vegetation remain green and physiologically active (principally, transpiring and fixing carbon, although stem diameter growth or leaf growth are also good indicators) during extended dry periods of the year?
14. Within a small region (and thus an area having the same annual rainfall, temperature and vapour pressure deficit) and in an area not having access to run-on or stream or river water, do some ecosystems show large seasonal changes in leaf area index whilst others do not?
15. Are seasonal changes in groundwater depth larger than can be accounted for by the sum of lateral flows and percolation to depth (that is, is vegetation a significant discharge path for groundwater; (Cook et al. 1998))? Clearly, if the error terms in the estimation of lateral flow and percolation to depth are of similar magnitude or greater than the rate of vegetation water, this method may not be appropriate.

Affirmative answers to one or more of these questions leads to the inference that the system is a GDE. However, this does not provide any information about the nature of the dependency (obligate or facultative) nor about the groundwater regime (e.g. timing of groundwater availability, volume utilised, location of surface expression, the pressure of the groundwater aquifer required to support the surface discharge of groundwater) needed to support the ecosystem.

13.3.2 Hydrological Indication of GDE Status

In shallow unconfined aquifers where roots of vegetation are directly accessing the water table (via the capillary zone usually), it is possible to discern the diurnal pattern of vegetation water-use in sub-daily fluctuations in the depth-to-groundwater (Gribovszki et al. 2010). Although diurnal changes in atmospheric pressure or temperature (which induce changes in water volume, evaporation and condensation) and inputs of rainfall can cause changes in groundwater depth, it is still possible to identify and sometimes quantify the extraction of groundwater through transpiration (Gribovszki et al. 2010).

White, in 1932, was possibly the first to use sub-daily changes in groundwater depth to quantify transpiration use of groundwater (White 1932). An idealised representation of the deil pattern of groundwater depth in a shallow unconfined aquifer is shown in Fig. 13.2.

The solid continuous oscillating curve represents the cycle of groundwater drawdown (because of *ET*) during the day followed by the rebound of the water table when *ET* returns to zero (assuming no nocturnal transpiration) at night. The dashed straight line (with slope $= r$) is used to estimate the amount of water

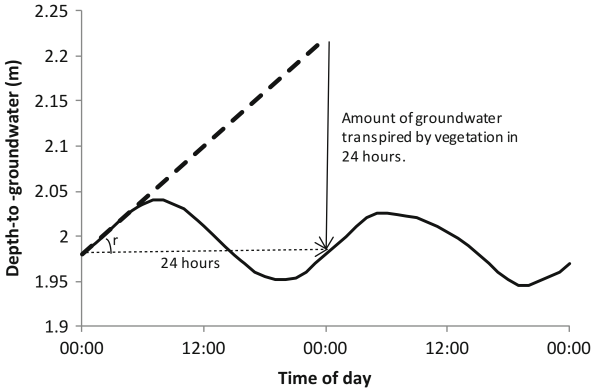


Fig. 13.2 A schematic representation of changes in depth-to-groundwater due to vegetation transpiration

transpired by vegetation in 24 h (0:00 h to 0:00 24 h later; indicated by the horizontal dotted arrow). This is represented by the vertical arrow which is the difference between the groundwater depth that would have occurred in the absence of vegetation water-use and the observed groundwater depth. By applying this methodology it is possible to identify the location of a GDE, thereby providing the first step in managing both the groundwater and the dependent ecology.

Lautz (2008) provides a detailed analysis of groundwater use using the White method of analyses of sub-daily changes in groundwater depth. She shows that spatial differences in groundwater use can be explained by differences in vegetation type (riparian wetland and grassland) and specific yield of the aquifer. As expected, the ratio of groundwater-to-soil water extraction increased as soil moisture content declined as a function of time since rain.

13.3.3 Geochemical Indication of GDE Status: Tracers and Isotopes

Geochemical studies, particularly isotopic analyses of water samples, can be used to distinguish groundwater sources from other water sources (e.g. atmospheric, soil water, or stream water sources), and used to identify source areas and groundwater residence time (e.g. Winograd et al. 1998; Monroe et al. 2005). Mineral deposition and helium isotope expression through groundwater discharge also can indicate groundwater discharge (Crossey and Karlstrom 2012), as attested to by the presence of certain plant species and invertebrates. For base flow systems (that is, rivers and streams showing significant flows during periods of zero surface or lateral flows), measurements of the chlorofluorocarbon, magnesium or radon concentrations of river and groundwater supply can identify and quantify the amount and timing of groundwater inflows into the river (Cook et al. 2003).

Stable isotopes (such as deuterium (^{2}H) and ^{18}O) can be used for these systems too, as can artificial labelling with tracers, such as lithium. When tracers are added to the groundwater, the subsequent uptake into vegetation is usually conclusive

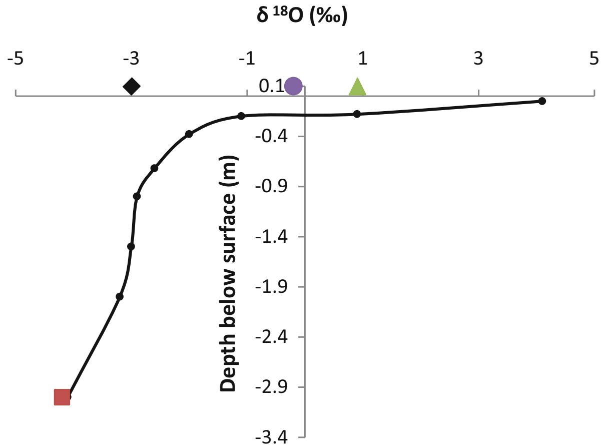


Fig. 13.3 An example of the use of ^{18}O analyses of xylem water, soil water and groundwater in a study of multiple species growing in northern Yucatan (Mexico). The ^{18}O content of soil declines with depth through the soil profile and eventually groundwater is reached (at 3 m; *brown square*). The xylem ^{18}O content of three species (*Ficus* spp. *green triangle*; *Spondias* spp. *purple circle*; and *Talisia* spp. *black diamond*) is also presented. *Ficus* was the least reliant on groundwater whilst *Talisia* was the most reliant (Redrawn from Querejeta et al. 2007)

proof that access by that vegetation is occurring. However, the presence of a tracer in a shallow rooted species can occur if neighbouring deep rooted species exhibit hydraulic lift and the shallow rooted plants then "harvest" this water (Caldwell et al. 1998). When a close match between groundwater isotope composition and xylem isotope composition is made, we can conclude that the vegetation is using groundwater.

Direct evidence that vegetation is using groundwater can be obtained by comparing the stable isotope composition of groundwater, soil water, surface water (where relevant) and vegetation xylem water (Kray et al. 2012; Lamontagne et al. 2005; O'Grady et al. 2006; Thorburn et al. 1993; Zencich et al. 2002; Spałek and Pro·ków 2011). A direct comparison of periodic measurements was made by Hunt et al. (1996) who showed that time integration provided by measurements of isotopic composition was a valuable tool that provide insights not available from non-isotopic techniques. Where sufficient variation in isotopic composition among these sources occurs then it is possible to identify the single or the most dominant source of water being used by different species at different times of year (Zencich et al. 2002). An example of the use of ^{18}O isotope analyses of xylem water, soil water and groundwater is shown in Fig. 13.3.

Mixed-member models are available that allow estimation of the relative contribution of multiple sources of water to the water absorbed by roots (Phillips and Gregg 2003; Kolb et al. 1997). Thus the use of stable isotopes can provide information about spatial and temporal variation in groundwater dependency and rates of groundwater use within and between species and ecosystems. Application of stable isotope analyses to quantify the rate of water use is discussed in Sect. 13.4.4.

13.3.4 Geomorphological Indicators of GDE Status

The various springs spheres of discharge (springs types) generate characteristic geomorphology and soils that may indicate groundwater dependence. Travertine mound-forming springs and hanging gardens are obvious examples of distinctive GDE geomorphology. Aerial photographic analysis of spring channels is commonly used to plan springs restoration projects (e.g. Ramstead et al. 2012). Because the geometry of springs channels is often erratic and non-sinuous (Griffiths et al. 2008), detection of such channel configuration is one indication of a spring flow domination, rather than surface flow domination (Springer et al. 2008). In hypocrenes, excavation of shallow wells or soil pits/cores can help identify groundwater sources, and among other springs types, discrete particle size arrays may result from constancy of discharge from some types of springs.

Geochemical deposits such as travertine commonly indicate groundwater dependence in mound-forming, hypocrene, geyser, and other springs types. Montezuma Well, the massive travertines along the Colorado River, and collapsed travertine mounds in the Tierra Amarilla region of northern New Mexico, are all examples of springs-related landforms (Crossey and Karlstrom 2012; Johnson et al. 2011; Newell et al. 2005).

In arid regions, organic soil development at springs can be extensive, distinctive, and dateable using radiocarbon techniques. Groundwater dependent peat deposits may be massive and can persist for millennia (e.g. Haynes 2008). Peat deposits more than 2 m thick were mined commercially in the Upper Carson Slough in Ash Meadows, a spring fed tributary of the upper Amargosa River basin in southern Nevada (McCracken 1992). If site geomorphology has not been much altered, these distinctive groundwater-generated landforms and soils features may remain identifiable, even if the aquifer has been largely dewatered.

13.3.5 Biotic Assemblages as GDE Status

Throughout the world, both in terrestrial and subaqueous settings, springs are widely known to support unique aquatic and wetland plant species and unique assemblages. In one of hundreds of examples of unusual springs-dependent plant species, Spałek and Pro·ków (2011) reported a highly isolated population of springs-dependent *Batrachium baudotii* (Ranunculaceae) in a karst spring in central Poland. The few remaining mound springs between Guildford and Muchea in Western Australia support restricted wetland graminoid plant assemblages, with Cyperaceae, Juncaceae, and Restionaceae, as well as flooded gum (*Eucalyptus rudis*) and bracken fern (*Pteridium esculentum*) (Blyth and English 1996).

In addition to springs-dependent aquatic and wetland species, the dendrochronology of trees from the periphery of springs also may be useful for establishing flow perenniality. Melis et al. (1996) used such data to evaluate flow variability of springfed Havasu Creek in Grand Canyon, reporting that the *Fraxinus velutina* cores revealed complacency of growth, indicating perennial flow over 80 years.

Surface-dwelling groundwater dependent species that indicate long-term groundwater flow perenniality include several groups of plants, invertebrates, fish, and amphibians. Among the plants in North America, such springs-dependent species are selected sedges (Caryophyllaceae), rushes (Juncaceae), and herbaceous taxa (e.g. some Primulaceae, *Toxicoscordion* spp., *Flaveria mcdougallii*). Among invertebrates, hydrobiid spring snails commonly are restricted to springs sources and channels, particularly the *Pyrgulopsis* and *Tryonia* (Hershler 1998, 2014), as are some members of the aquatic beetle families *Elmidae* and *Dryopidae* (Shepard 1993). In our studies of montane springs in the American Southwest, chloroperlid stoneflies and turbellarian flatworms are often springs-dependent species in cool-cold natural waters. Among North American fish, the pupfishes (*Cyprinodontidae*) and goodeid topminnows (*Goodeidae*) are often springs-dependent, and often are tightly restricted to individual springs (e.g. Minckley and Deacon 1991; Unmack and Minckley 2008). Among southwestern amphibians, populations of native ranid frogs in the genus *Lithobates* (*Rana*) are often associated with groundwater dependent wet meadows (cienegas, GDE fens). The giant aquatic hellbender salamander, *Cryptobranchus alleganiensis bishopi* only occurs in clear water springfed stream segments in the Ozarks. Several turtle species in eastern North America hibernate on the periphery of coldwater springs, where they are cooled but are protected from freezing (Nickerson and Mays 1973; Ernst and Lovich 2009).

13.3.6 Historical Documentation of GDE Status

Historical documentation is often useful for establishing GDE status and the perenniality of springs flow. Many sources of historical information may be available for such documentation, such as historical photographs and diaries, and interviews with long-term stewards and community elders. Such historical information can be quite valuable for understanding change through time; however, locating, determining the validity of such information, and compiling and interpreting the information can be challenging.

13.3.7 Remote Sensing

Detection of GDEs through remote sensing (RS) includes the use of infrared and other aerial thermal imaging, and has been used successfully to locate groundwater sources, particularly during seasons with the greatest temperature differences between air and groundwater temperatures. Remote sensing (RS) provides a rapid and spatially extensive technique to assess vegetation structure (e.g. leaf area index, basal area), vegetation function (e.g. canopy temperature, rates of evapotranspiration and "greenness") and relationships amongst climate variables, vegetation function and vegetation structure.

An underlying conceptual model for the application of RS to identifying the location of GDEs has been that of "green islands". In this approach, the structure or function of one pixel in a RS image is compared to that of an adjacent pixel. If a GDE covers a significant fraction of the area of one pixel but not the other, it is assumed that during prolonged dry periods the structure/function of the two vegetation types will diverge. This is because the vegetation accessing groundwater is not experiencing soil dryness to the same extent (if at all) as the vegetation that is not accessing groundwater. Under the green islands conceptual model, assessments of vegetation *structure* or *function* are determined for the site of interest and compared to adjacent "control" sites, either at a single time, or preferentially, across several contrasting times (comparisons across "wet" and "dry" periods usually).

In the United States, aerial thermography surveys of the largest of Florida's springs, Silver Springs, were conducted along the spring-fed run out channel and detected new spring orifices over 1200 m below the first source (Munch et al. 2006). Remote sensing techniques can be successfully used in low-gradient terrain that is not covered by dense vegetation. The U.S. Forest Service conducted remote sensing analysis for fens in the Rocky Mountains to detect fens (U.S. Forest Service 2012), reporting good success in locating large fens that were exposed. However, a similar remote sensing effort in the topographically complex Spring Mountains of southern Nevada detected fewer than 50 % of the more than 200 springs in that range (U.S. Forest Service 2012).

13.3.7.1 Application of Vegetation Indices Derived from RS

Münch and Conrad (2007) examined three catchment areas in the northern Sandveld of South Africa. They used Landsat imagery to identify the presence/absence of wetlands and combined this with GIS terrain modelling to determine whether GDEs could be identified using a landscape "wetness potential". It is important to note that this application focused on Class II GDEs – those reliant on a surface expression of groundwater. They applied the "green island" philosophy and compared the attributes of potential GDEs with the attributes of surrounding land covers at three contrasting times: July when rains started at the end of a dry year, August, in the winter of a wet year and at the end of a dry summer. They concluded that RS data could be used to classify landscapes and when this was combined with a spatial GIS based model using landscape characteristics they could produce a regional-scale map of the distributions of GDEs. However, it is not known whether this approach could be applied to Class III GDEs (those reliant on sub-surface access to groundwater).

In arid and semi-arid regions, plant density is often correlated with water availability. When groundwater is available to vegetation, plant density tends to be larger than adjacent areas where groundwater is unavailable. Lv et al. (2012) used remotely sensed images of a vegetation index (the Normalised Difference

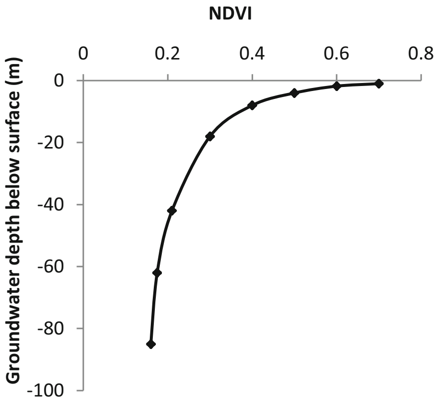


Fig. 13.4 The relationship between NDVI and depth-to-groundwater for the Hailiutu River catchment in northern China (Redrawn from Lv et al. 2012)

Vegetation Index; NDVI) to assess changes in NDVI as a function of depth-to-groundwater in northern China. A 25 m resolution digital elevation model and groundwater bore data were used to generate a contour map of groundwater depths across the 2600 km^2 catchment. Approximately 29,000 pixels of 300 m resolution of NDVI data were then used and the following relationship determined (Fig. 13.4):

This study demonstrated that the largest NDVI, a reliable measure of vegetation cover, occurred at the shallowest depths of groundwater and that cover declines curvilinearly with increasing depth-to-groundwater. They further analysed NDVI data and identified five land classes, including water bodies and bare earth as one land class, having a zero vegetation cover; and farmland and riparian zones as another class having the largest NDVI. The remaining three classes had intermediate values of NDVI. They then showed that the vegetated classes exhibited different responses to depth-to-groundwater. A cut-off of approximately 10 m depth-to-groundwater was apparent; when the water table was lower than 10 m, vegetation cover was insensitive to further increase in groundwater depth.

A similar method was applied by Jin et al. (2011) for the Ejina area in NW China. Despite much of the region being within the Gobi desert, with approximately 40 mm annual rainfall, an oasis located in the northern part of Ejina supports extensive agricultural and native vegetation. The NDVI was used by Jin and co-workers, along with 13 groundwater bores, from which relationships between NDVI and groundwater depth for three vegetation classes (grassland, woodland and scrubland) were established. Surprisingly, maximum NDVI were not observed at the shallowest groundwater sites for any vegetation class but at intermediate (2.5 – 3.5 m) depths. A cut-off of 4.4 m depth-to-groundwater was observed such that vegetation was absent in regions where groundwater depth exceeded 5.5 m.

Dresel et al. (2010) used geological, hydrogeological and ecological data to define regions having common physical and climatic profiles and which therefore should have similar RS signals. MODIS eVI and Landsat NDVI data were used and aridity thresholds (calculated as the Thornthwaite index) for individual regions developed based on a correlation analysis of Landsat summer NDVI images and

MODIS eVI. Both of these are surrogate measures of productivity, with eVI generally performing better than NDVI (Campos et al. 2013).

Three methodologies were applied by Dresel et al. (2010). In the first, the MODIS eVI images identified pixels with a consistent photosynthetic activity throughout the year and pixels having variation across the year that was less than one standard deviation of the mean were deemed to show consistent productivity all year. For the second method, Landsat NDVI images were used to identify areas with contrasting photosynthetic activity for a wet year and a dry year. In the third method, an unsupervised classification of Landsat spectral data was used to identify spectral signatures of pixels that were deemed to be highly likely to use groundwater using expert local knowledge and then find other pixels with similar spectral signatures. Species specific differences in spectral signatures have been identified previously (Nagler et al. 2004). By combining all three methods within a GIS and finding pixels with a consistent productivity all year plus a high contrast between other local pixels plus a similar spectral signature to known GDEs, it was possible to identify all pixels across a catchment that had a very high probability of being a GDE. Ground truthing was then required.

An alternate approach to mapping the location of GDEs involves mapping of discharge zones, especially discharge through transpiration of vegetation and discharge to the ground surface. Discharge of groundwater to the surface (to swamps, wetlands and rivers) or through transpiration exerts a profound effect on the ecology of those systems utilising groundwater. To define the spatial extent of discharge across a landscape requires a multi-disciplinary approach that incorporates knowledge of geology, hydrology, ecology and climate (Tweed et al. 2007). Leblanc et al. (2003a, b) for example, used thermal, Landsat optical and MODIS NDVI data coupled to digital elevation models and depth-to-groundwater data to locate discharge areas in a large semi-arid basin in the Lake Chad basin in Africa. Tweed et al. (2007) examined discharge (and recharge) of the Glenelg-Hopkins catchment of southeast Australia. Discharge occurred through direct evaporation of the water table, with a likely limit of 5 m depth from which evaporation could occur; transpiration by vegetation from regions overlying a shallow unconfined aquifer and discharge to the ground surface to localised depression, break-of-slope localities and to wetlands, rivers and the ocean. The methodology they employed is summarised thus (from Tweed et al. 2007, Fig. 13.5).

Key indicators of groundwater discharge used in this study include:

1. Low variability of vegetation activity across wet and dry periods (seasons or years) using the NDVI as a measure of vegetation photosynthetic activity.
2. Topographic depressions and breaks of slope across the catchment, derived from a digital elevation model for the catchment to identify potential locations for surface discharge. A topographic wetness index (w) was calculated from: $w = \ln(1/\tan\beta)$ where β is the gradient of the slope of the land surface. Identification of concave slopes by identifying negative second-derivatives of slopes was used to identify areas where potential zones of saturation (arising from groundwater discharge) may occur across the landscape.

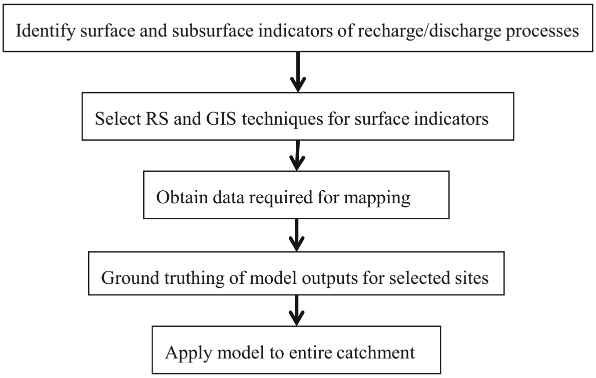


Fig. 13.5 A schematic of the methodology used by Tweed et al. (2007) in the use of RS and GIS to identify the location of GDEs in a landscape

3. Groundwater depth data were used to produce a groundwater flow and these were combined with the digital elevation map to produce a depth-to-groundwater map.

From this approach a detailed map of potential discharge zones across the entire 11,000 + km^2 catchment was produced that far exceeded the ability if only the limited bore data had been used. A map of the standard deviation of the NDVI was able to identify locations where groundwater was supporting vegetation activity and thus identify GDEs across the catchment. A potential limitation to this method was that it tended to be most accurate in drier parts of the catchment where rainfall is more likely to limit vegetation activity. It was also found that identification of topographic depressions was a more reliable indicator for groundwater discharge than identification of break-of-slope.

13.3.7.2 RS Derived Estimates of Water Fluxes

The energy balance equation for land surfaces can be written thus: $LE + H = R_n - G$, where LE is latent energy flux (=ET), H is sensible heat flux, R_n is net radiation and G is soil heat flux. Differences in temperature between boundary air temperature and canopy temperature can be used to estimate sensible heat flux. Assuming over a 24 h cycle $G = 0$, and R_n is either measured or derived from remote sensing data, then LE (that is, ET) is calculated by difference. Li and Lyons (1999) used three models based on surface temperatures to estimate ET. The first model only used differences in surface and air temperature to calculate ET, the second model required NDVI data and surface temperature. This model requires the four extreme values of surface temperature and NDVI to be present within the area of study (i.e. patches of dry bare soils, wet bare soil, wet fully vegetated patches and dry (water stressed) fully vegetated surfaces). This makes its application problematic. The third method simply used the Priestley-Taylor equation (see Li and Lyons 1999) to estimate potential ET (E_p).

Two of the key functional attributes of terrestrial ecosystems are the rates of water-use (either transpiration or evapotranspiration) and the rates of carbon fixation. Fluxes of transpired water and carbon uptake are coupled through the action of stomata, through which both gases must flow. It is because of the tight coupling of water and carbon fluxes that vegetation indices such as NDVI or eVI, which are good proxies of productivity and hence carbon flux, can be successfully applied in looking for GDEs, where it is an increase in water supply that drives their structural and functional differences (compared to adjacent no-GDEs).

13.3.8 GDE Mapping and Database Challenges

Information management constitutes a serious challenge for understanding and managing GDEs. Accurately georeferencing and archiving data on the distribution and ecohydrology of springs and other GDEs first involves developing a suitable database framework (Springs Stewardship Institute 2012). Some or many of the above methods for determining GDE distribution allows development of a geographic information system georeferenced map of springs within landscapes. However, a common problem in such mapping efforts is resolution of duplication error. We have repeatedly found that: (a) no single source of information (usually GIS layers or survey reports) provides a complete list of springs or other GDEs within a large landscape; (b) that each information source contains unique springs not found elsewhere; and (c) that the same GDEs may be mapped in multiple places with different names. Stevens and Ledbetter (2012) used 10 sources of information to identify 150 springs on the North Kaibab Forest District of northern Arizona, 50 % more springs than had been documented by the managing agency, and field surveys increased the number of known springs in that landscape to more than 200. Development of an adequate map and database on the springs of large landscapes provides an essential tool for monitoring, modelling and further research on the status of the underlying aquifers.

13.4 Estimating Rates of Groundwater Use by Class III GDEs

Estimating groundwater needed to maintain GDE function is an essential step to the sustainable management of both GDEs and groundwater resources. However, it poses many methodological impediments, including:

1. Up-scaling from tree-scale measurements of tree water-use;
2. Partitioning total vegetation water-use into rain and groundwater sources;
3. Understanding seasonal/life-cycle variations in the rates of groundwater use;
4. Understanding the influence of climate at inter-annual time-scales on rates of tree water-use and the partitioning of water-use into rain and groundwater sources.

Moreover, what is required for the establishment and persistence of GDE function is often not well characterized; therefore the emphasis has been on

Table 13.1 Three methods to estimate rates of groundwater discharge through vegetation in data poor areas, summarised from Leaney et al. (2011)

Model	Input data	Method
Groundwater risk model	Climatic characterisation (rainfall, evaporation), depth-to-groundwater, soil profile characterisation (depth, texture, moisture holding), groundwater salinity	Uses a simple water balance approach to estimate the probability of groundwater use and estimate groundwater discharge
Ecological optimality model	Climatic characterisation (rainfall, evaporation), long term average Leaf Area Index (LAI)	Estimates groundwater discharge based on the difference between LAI of GDE and theoretical LAI for a given climate wetness index (P/E_0)
Groundwater discharge – salinity function	Groundwater salinity	Estimates groundwater discharge based on empirical relationship between groundwater discharge and groundwater salinity

measuring water use in existing GDEs and using this characterization as a basis for baseline conditions. A range of tools are available to estimate groundwater use by Class III GDEs. These are now briefly discussed.

13.4.1 A Spreadsheet Tool

Because of the paucity of data on points (1)–(4) above, Leaney and co-workers developed a novel, simple, but useful first-order method to estimate groundwater use of vegetation using a simple excel spreadsheet tool (Leaney et al. 2011). The excel spreadsheet includes three methods to estimate rates of groundwater discharge through vegetation:

(a) a groundwater risk model;
(b) an ecological optimality model; and
(c) a groundwater discharge salinity function.

These are summarised in Table 13.1.

The groundwater risk model is a simple water balance model that uses historical monthly rainfall and monthly evaporation data for any site. The soil profile is defined by the user and soil texture is used to estimate soil moisture characteristics for each layer. Groundwater discharge through vegetation is deemed to occur whenever evapotranspiration (*ET*) exceeds rainfall plus the soil water stores.

13.4.2 Sub-daily Fluctuation in Groundwater Depth

In addition to being used to identify the location/presence of a GDE in a landscape, the White method (White 1932) described in Sect. 13.3.2 for analysing sub-daily changes in depth-to-groundwater can be used to quantify rates of groundwater use. The volume of water transpired is calculated from the change in volume of water in the aquifer that would account for the observed changes in the depth of the water table on an hourly or daily basis, assuming the specific yield of the aquifer is known with sufficient accuracy and confidence. Butler et al. (2007) examined the controls of variation in rates of groundwater use across several riparian sites in the High Plains region of the USA. They found that the principle drivers of vegetation water use were meteorological, vegetation attributes and the specific yield of the aquifer. Their estimates of groundwater use (3–5 mm d^{-1}) agreed well with estimates derived from sapflow measurements of tree water use. For a detailed assessment of the technical problems inherent in application of the White method, the reader is referred to Loheide et al. (2005). Further examples of estimating rates of groundwater use using the White method can be found in Lautz (2008), Martinet et al. (2009) and Gribovszki et al. (2008).

13.4.3 Using Remote Sensing to Estimate Groundwater Use

Methods for remotely sensed estimates of groundwater discharge are being developed. It is important to quantify the water balance of arid and semi-arid groundwater basins to define safe yields for those resources. Obtaining accurate and spatially distributed estimates of discharge through vegetation is problematic, expensive and time consuming using field techniques. Consequently, Groeneveld and Baugh (2007) derived a new formulation of the standard NDVI which stretches the NDVI distribution for vegetation from zero to one. This new NDVI (*NDVI**) can be calibrated to quantify actual rates of evapotranspiration (ET_a) and the calibration only requires standard weather data from which to calculate (E_o) (the grass reference ET calculated using the Penman-Monteith equation, as described in the FAO-56 method (Allen et al. 1998). The *NDVI** is functionally equivalent to the crop coefficient (K_c) commonly used in micrometeorology. This methodology is especially applicable to vegetated arid and semi-arid sites with a shallow water table where rainfall is low, often erratic but water supply to roots is relatively constant. Consequently ET closely tracks ET_o, which varies as a function of solar radiation, wind speed and vapour pressure deficit.

Groeneveld et al. (2007) applied the *NDVI** methodology to three disparate arid sites in the USA where annual ET_a values were available through use of Bowen ratio or eddy covariance equipment. A linear correlation ($R^2 = 0.94$) between measured annual ET_a and mid-summer *NDVI** was obtained across the pooled, three-site data, despite very different vegetation composition and structure across the three sites.

Deducting the contribution of annual rainfall to annual ET_a yields the amount of groundwater that is transpired by the vegetation (ET_{gw}). Thus, $ET_{gw} = (ET_o - rainfall)NDVI^*$ Across sites and across years, the average error in ET_{gw} was estimated to be about 12 %, which in the absence of field assessments is a very valuable estimate of groundwater use.

Groeneveld (2008) applied the methodology of Groeneveld et al. (2007), using mid-summer NDVI data to estimate annual total ET of alkali scrub vegetation in Colorado. An estimate of annual groundwater use was then estimated as the difference between annual rainfall and annual ET for each year. On-site estimates of groundwater use were larger than those estimated using NDVI data and ET_o because the remote sensing method does not include surface evaporation of groundwater. Annual ET_{gw}* were compared to measurements made by Cooper et al. (2006) at the same site agreed to within 20 %. Similarly, as noted earlier in the discussion of RS methods to find ET, Scott et al. (2008) developed a numeric relationship for ET_a and concluded that the difference between ET_a and annual rainfall was groundwater use.

13.4.4 Using Stable Isotopes to Estimate Rates of Groundwater Use

Stable isotopes have been used extensively to provide estimates of the proportion of total vegetation water use that is derived from groundwater (Feikema et al. 2010; Kray et al. 2012; Máguas et al. 2011; McLendon et al. 2008; Querejeta et al. 2007). Thus, an independent estimate of rates of water use are required in addition to analyses of the stable isotope composition of soil water, groundwater and xylem water. Methods to estimate rates of vegetation water use include eddy covariance (Eamus et al. 2013), measurement of rates of sapflow (Zeppel et al. 2008) and remotely sensed estimates (Nagler et al. 2009). When only a single isotope is analysed (^2H or ^{18}O) a linear mixing model can distinguish between only two potential sources of water (groundwater and soil water). If both isotopes are used, spatial resolution is increased and one can distinguish between three sources of water, but only if the two isotopic compositions are independent of each other, which is often not the case. Interestingly, early work in 1996 established that the application of stable isotope analyses was found to be the most accurate method available in a comparative analysis of wetland groundwater inflows (Springs Stewardship Institute 2012).

Two generalities can be identified in the results of stable isotope studies of GDEs. First, as depth-to-groundwater increases, the proportion of total vegetation water-use that is derived from groundwater diminishes (O'Grady et al. 2006) although this can vary amongst different vegetation communities (McLendon et al. 2008). Second, the proportion of groundwater used by vegetation usually (McLendon et al. 2008) but not always (Kray et al. 2012) increases as time since last rain increases and soils dry out and thus seasonality of groundwater use may occur when rainfall is highly seasonal and groundwater availability is maintained throughout the dry season (O'Grady et al. 2006).

Stable isotope composition varies as a function of depth (Fig. 13.3) and taking an average value to represent the entire rooting depth of the vegetation leads to errors. Even with two independent isotopes available for analyses, the relative contribution of only three sources can be determined. To overcome this limitation, Cook and O'Grady (2006) developed a simple model of water uptake whereby the relative uptake from different depths is determined by (1) the gradient in water potential between the soil and the canopy; (2) root distribution as a function of depth; and (3) a lumped hydraulic conductance parameter. Isotopic composition of water through the soil profile and of xylem water is then used to constrain root distributions (as opposed to measuring this destructively *in situ*). This model has several advantages over the more commonly used end-member (Phillips and Gregg 2003) analyses: (1) produces a more quantitative estimation of proportion of water extracted from different depths (including groundwater); (2) does not require distinct values of isotope composition for end-member analyses and therefore can deal with the more typical grading of isotope composition observed through the soil profile; and (3) is based on simple ecophysiological principles. Sapflow sensors were used to measure rates of tree water use across four species growing in a tropical remnant native woodland and this was up-scaled using plot basal area. Cook and O'Grady (2006) demonstrated that two species were sourcing 7–15 % of its transpirational water from the water table, a third species was accessing 100 % of its water from the water table and a fourth species was accessing between 53 % and 77 % of its water from the water table—further confirmation of niche separation of patterns of water uptake for co-occurring species.

13.5 Threats to GDEs

Human activities threaten GDEs by disturbing habitats, depleting groundwater reserves, altering the groundwater regime at a site beyond the natural bounds of variation previously experienced at that site, and degrading groundwater quality. Globally, GDEs are and will continue to be threatened by groundwater depletion due to increasing water demands from growing populations and increased industrial demand (Danielopol et al. 2003). Wada et al. (2010) estimated that global groundwater depletion (i.e. groundwater abstraction in excess of recharge) in sub-humid to arid areas was approximately 280 $km^3\ yr^{-1}$ in 2000, doubled from 1960. Increasing water demands was projected to greatly outweigh climate change in defining global water resource to 2025 (Vörösmarty et al. 2000). Locally, human activities have impacted GDE habitats through vegetation clearing, filling or draining of wetlands and alteration of surface water courses. Regionally, major anthropogenic threats to GDEs include

- alteration of surface water regime and quality through river regulation and land-use change;

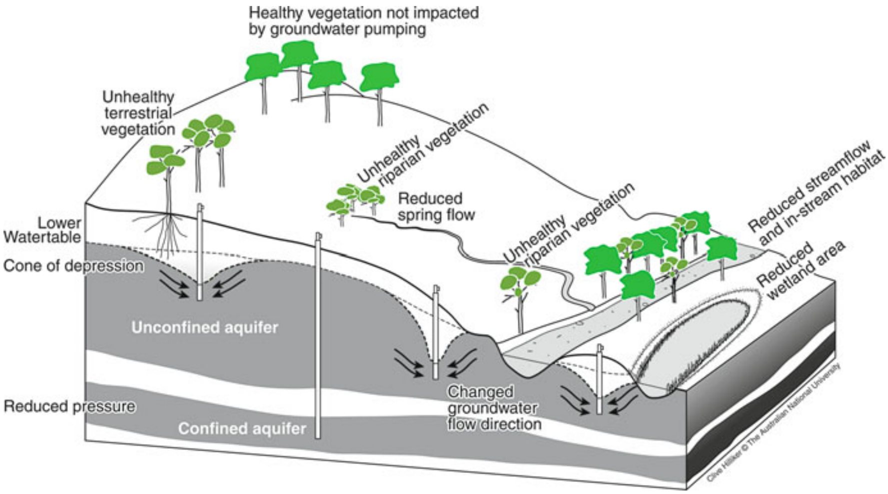


Fig. 13.6 Diagram showing the potential impacts of groundwater pumping on GDEs

- alteration of groundwater regime and quality as a result of agricultural practices, urban and industrial development, mining activities and plantation forestry (Fig. 13.1).

For GDEs that rely on both surface and groundwater sources, surface water regime (including flooding) and quality are considered the most important factor threatening GDEs (Eamus et al. 2006). Evidences of ecosystem change due to flow alteration and surface water quality decline have been reviewed elsewhere (Nilsson et al. 2005; DeFries et al. 2004). This section focuses on groundwater regime and groundwater quality.

13.5.1 Anthropogenic Threats to Groundwater Regime

Groundwater extraction is one of the major threats that alters groundwater regime. Groundwater has been extracted to support agricultural activities (especially irrigation), to satisfy residential water-use and to support urban and industrial development. In these cases, groundwater is often extracted through pumping wells in confined or unconfined aquifers. Excessive groundwater pumping in a confined aquifer will depressurise the entire confined aquifer and reduce groundwater discharge to springs (Weber and Perry 2006) (Fig. 13.6). The impact is at a regional scale. In contrast, impact of groundwater pumping from an unconfined aquifer is more localised. In unconfined aquifers, when extraction is faster than recharge, groundwater depth increases forming a "cone of depression" around the well that can extend for many hundreds of meters from the well (Fig. 13.6). In addition, groundwater flow direction can be changed because of the generation of new hydraulic gradients: groundwater may no longer flow into the local stream, and some water may be drawn from the stream to the well, thereby reducing stream flow. The time lag between extraction and a reduction in discharge to a stream vary from a few hours to many centuries, depending on extraction locations (relative to the stream), extraction volume and groundwater flux (Evans 2007).

Increased depth-to-groundwater and the disappearance of springs have been reported around the world and are associated with excessive groundwater pumping for agricultural and urban development, mining activities and plantation forestry (Fig. 13.1). Depth-to-groundwater has increased by 4–17 m in an irrigation region of northwest China, forming several cones of depression covering about 1000 km^2 (Wang et al. 2003). Similarly, Burri and Petitta (2004) observed progressive disappearance of numerous springs in the Fucino Plain, Italy, due to increased agricultural water-use for water-intensive horticultural crops and second harvest practices. In some areas of extensive urban development, groundwater depletion has occurred at alarming rates. For example, in London the water table has dropped more than 70 m below the surface (Elliot et al. 1999); in Bangkok, the water table has dropped by 25 m since 1958; in Tamil Nadu, India, a 30 m decline in 15 years has occurred (Danielopol et al. 2003). Muñoz-Reinoso (2001) reported that the decline of water table in Doñana, Spain was primarily due to pumping for urban water supply of a tourist resort and secondarily due to the transpiration of large pine plantations. Mine dewatering (removal of water by pumping or evaporation) can have large impacts on aquifer and cave system locally, and springs close to mine sites. Cluster of mining operations can impact depth-to-groundwater at regional scales due to their cumulative effects (Clifton and Evans 2001).

In addition to groundwater extraction and mine dewatering activities, in-channel gravel or sand mining can cause the incision of a riverbed which lowers the alluvial water tables (Kondolf 1994). Scott et al. (1999) reported water table declines of more than 1 m at sites affected by gravel mining (compared to no significant decline at control sites). Sustained lowering of the water table greater than 1 m has led to significant declines in *Populus* growth and 88 % mortality over a 3-year period (Scott et al. 1999). Water-logging, typically caused by forest clearing and poorly managed irrigation in agricultural lands can result in a rise in the water table, and associated impacts through impaired root function because of the development of anoxic conditions within the root zone (Pimentel et al. 1997).

13.5.2 Anthropogenic Threats to Groundwater Quality

Reports of groundwater contamination caused by human activities are abundant. Nitrate leaching from agricultural lands to shallow groundwater has been reported in many regions around the world (Andrade and Stigter 2009). Elevated nitrate levels in groundwater can be sourced from nitrogen fertilizers and manure, oxidation of organically bound nitrogen in soils, cattle feed lots, septic tanks and sewage discharge. Severity of contamination is modified by other factors such as lithology, dissolved oxygen levels and land-use. Andrade and Stigter (2009) reported that rice fields on fine-grained alluvium generally have low dissolved oxygen and minimal nitrate concentrations in groundwater due to denitrification. In contrast, areas with vegetable crops coupled with coarse grain lithology and high hydraulic conductivity have higher concentrations of nitrate in shallow groundwater. Discharge of nitrate enriched groundwater can alter nitrogen concentrations in the receiving water and hence increase the risk of eutrophication and algal blooms.

Pesticide contamination can be a problem for shallow groundwater. In the US, more than half of the wells in agricultural and urban areas contain one or more pesticide compounds (Gilliom et al. 2006). Using poor quality pesticides with low degradation rates, incorrect application of pesticides and inappropriate disposal methods can all lead to groundwater being contaminated by pesticides, among which herbicides are the most frequently detected in groundwater (Andrade and Stigter 2009).

Urban development can impair groundwater quality, thereby damaging urban ecosystems. Examples include leakage from septic tanks, underground fuel tanks, landfills, and use of fertilisers and pesticides for gardens and recreation areas. Animal rearing, horticultural activity, solid waste dumping, pit latrine construction and stormwater/sewage disposal have led to increased localised microbial and organic contamination of shallow groundwater (Kulabako et al. 2007; Massone et al. 1998). Foppen (2002) reported increased concentrations of almost all major cations and anions and acidification of groundwater at Sana'a, Yemen, due to continuous infiltration of wastewater into the aquifers via cesspits. More recently, urban groundwater in cities of Germany has been shown to be polluted with xenobiotics such as pharmaceuticals, personal care products (collectively known as PPCPs) and endocrine-active substances (Schirmer et al. 2011). However, their potential long-term effects on ecosystems and humans remain largely unknown.

Mining can contaminate groundwater during mining operation (e.g. leakage from tailings dams and crushed rock waste dumps, which can cover hundreds of hectares at a mine site), as well as the recovering phases after mine sites are abandoned (Younger and Wolkersdorfer 2004; Gao et al. 2011). Dewatering disturbs groundwater stratification, thereby altering the environment required by cave or aquifer ecosystems and associated stygofauna. Cidu et al. (2001) reported that mine closure and associated cessation of groundwater pumping and mine flooding may pose a contamination risk to shallow aquifers due to the rise of deep saline groundwater. Progressive mine flooding also causes groundwater contamination via weathering of ore minerals and remobilization of metals in the mine waste (Razowska 2001).

In summary, groundwater regime and quality are threatened by many human activities, including agricultural practices, urban and industrial development, mining activities and plantation forestry. These threats can have profound impact on GDEs in the short and long term, at local and regional scales. The impacts of groundwater abstraction on GDEs and their restoration are discussed below using two case studies.

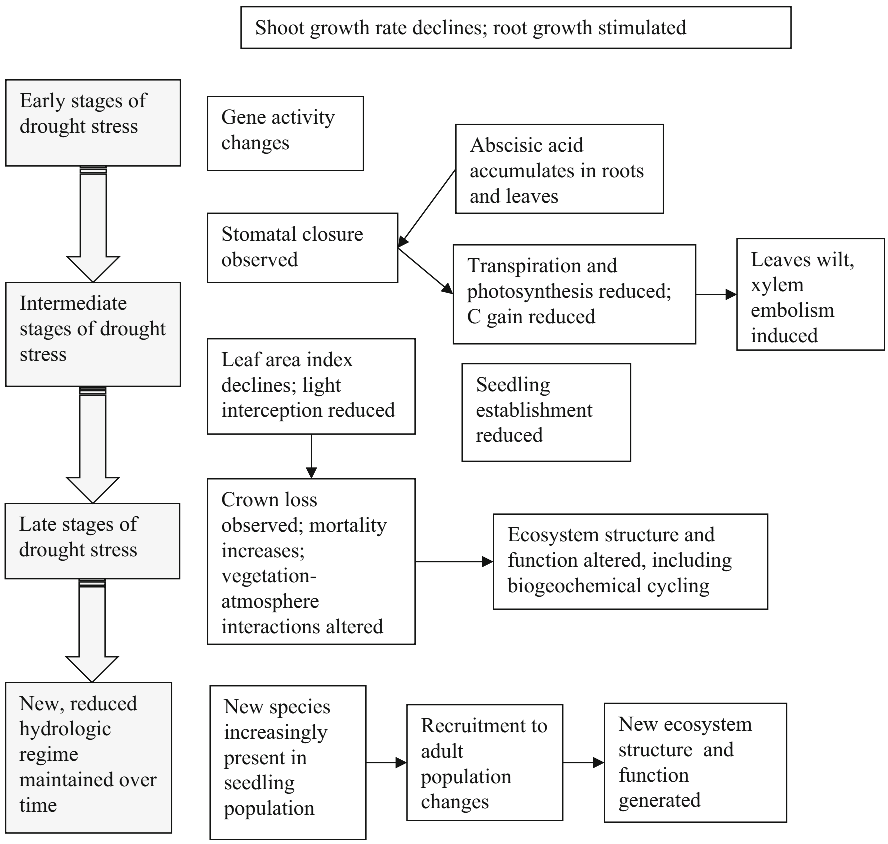


Fig. 13.7 Schematic outline of some of the changes in plant physiology, ecophysiology and ecology associated with short-, medium- and long-term changes in water availability

13.5.3 Case Study 1: Terrestrial Vegetation

The impacts of groundwater abstraction on woodlands has been documented for the Gnangara Mound, a shallow unconfined aquifer of the Swan Coastal Plain in Western Australia (Canham et al. 2009, 2012; Groom et al. 2000; Stock et al. 2012). Increased depth-to-groundwater is the result of a long-term decline in annual rainfall across the region, increased abstraction for human use and increased discharge (reduced recharge) arising from the development of a plantation industry in the region. A range of changes in plant physiology, ecophysiology and ecology are found associated with short-, medium- and long-term changes in water availability (Fig. 13.7).

In 1985 increased rates of summer abstraction in this Mediterranean climate resulted in increased and widespread mortality (up to 80 % mortality close to the abstraction bores) of the native *Banksia* woodland. To determine longer-term floristic changes arising from groundwater abstraction, a series of transect studies were initiated in 1988. A 2.2 m increase in depth-to-groundwater, coupled to higher-than-normal summer temperatures resulted in a 20–80 % adult mortality of overstory species and up to 64 % mortality in the understory species, 2 years after the start of groundwater pumping (Groom et al. 2000). Control sites, not impacted by groundwater pumping, did not display increased mortality.

Because of the large inter-species differences in rates of mortality, a further study examined the vulnerability of different species to reduced water availability (Canham et al. 2009; Froend and Drake 2006). Using xylem embolism vulnerability curves as an indicator of sensitivity to water stress, Froend and Drake (2006) compared three *Bankisa* and one *Melaleuca* species. They found that xylem vulnerability reflected the broad ecohydrological distribution of the species across the topographic gradient present at the site and they were able to identify a threshold leaf water potential below which increased mortality was likely.

Similarly, Canham et al. (2009) examined Huber values (the ratio of sapwood to leaf area), leaf-specific hydraulic conductivity (k_l) and xylem vulnerability of two obligate phreatophytes and two facultative phreatophytes. At sites were water availability was high (no increase in depth-to-groundwater) there were no interspecific differences in vulnerability to water stress. However, in a comparison of the upper and lower slopes (corresponding to larger and smaller depth-to-groundwater respectively) the two facultative phreatophytes (but not the obligate phreatophytes) were more resistant to xylem embolism at the upper slope than the lower slope, whilst one of the obligate phreatophytes did not alter its sensitivity (Canham et al. 2009).

In addition to differences in sensitivity of above-ground tissues to changes in water availability, it is likely that differences in the responses of root to changes in depth-to-groundwater contribute to the impact of changes in depth-to-groundwater on vegetation in GDEs. In a comparative study on two *Banksia* tree species, Canham et al. (2012) observed that root growth at sites with shallow depth-to-groundwater was in synchrony with above-ground growth patterns. This was in contrast to patterns observed at depth, where root growth occurred all year and was independent of aerial climate. As depth-to-groundwater increased during the summer in this winter rainfall site, roots grew increasingly deeper, following the capillary fringe. As recharge occurred in the winter and depth-to-groundwater declined, anoxia resulted in root death at depth. These authors concluded that the ability to rapidly increase root depth during the summer is a critical attribute of phreatophytes occupying sites with seasonally dynamic depth-to-groundwater.

Long-term (>2 years) studies of the influence of changes in depth-to-groundwater are relatively rare, despite the importance of such studies to the development of ecosystem response trajectories for the impact of groundwater abstraction. Froend and Sommer (2010) examined a rare, 40 year duration, vegetation survey data-set for the Gnangarra Mound in Western Australia. Although the long-term (1976–2008) average rainfall in 850 mm, this has been declining for the past 40 years. Currently the annual average is about 730 mm. This, along with increased groundwater abstraction, has resulted in increases in the depth-to-groundwater over the past 50 years of about 1 m. Seasonally, depth-to-groundwater fluctuates about 0.5–3 m, with a maximum depth occurring at the end of the summer. Two transects were used – a "control" transect where gradual increases in depth-to-groundwater (9 cm y^{-1}) have occurred as a result of the decline in annual rainfall over the past several decades; and an "experimental" transect where large rates of increase in depth-to-groundwater (50 cm y^{-1}) because of declining rainfall and extensive abstraction of groundwater have occurred. Three vegetation communities were identified with principal coordinate analyses and these were clearly associated with down-slope, mid-slope and upper-slope positions, corresponding to shallow, intermediate and deep depth-to-groundwater respectively. Species known to have a high dependency on consistent water supplies (mesic species) were dominant at the down-slope site whilst xeric species dominated the upper-slope sites.

On the control transect (slow rates of increase in depth-to-groundwater), the hypothesis that groundwater water abstraction would result in a replacement of the mesic by the xeric species was not supported. Most of the compositional and structural attributes of the three communities were unchanged. The principle community-scale response was a change in the abundance of mesic and xeric species rather than a complete replacement of one species for another. In contrast to the results of Shafroth et al. (2000), mesic species growing on sites with shallow groundwater were not more sensitive to increases in depth-to-groundwater than xeric species.

On the "experimental" transect where the increase in depth-to-groundwater was much faster (50 cm y^{-1}) changes in composition were far more pronounced and mass mortality observed across all classes (mesic to xeric) species. This result emphasises the importance of the rate of increase in depth-to-groundwater in determining the response of species and communities.

13.5.4 Case Study 2: Restoration of Springs

A systematic review of the literature of the restoration of arid-land springs was conducted by Stacey et al. (2011) to determine how successful projects were in restoring hydrology, geomorphology, and biological assemblage composition and structure in relation to those at natural springs with minimal anthropogenic disturbances. Unfortunately, the great inconsistency in the rationale for and in the implementation, monitoring, and reporting of springs restoration efforts globally made it impossible to conduct meta-statistical analyses of the quality of restoration. Stacey et al. (2011) recommended standardised ecosystem condition and restoration assessment protocols are needed to more clearly understand the success of projects. Because of the inability to report on a global summary of the success of restoration and management, we provide a case study by specific spheres of discharge to provide some lessons learned from restoration and management actions.

Hoxworth Springs is a rheochrene spring on the Mogollon Rim of the southwestern Colorado Plateau (Godwin 2004). This system is typical in both the morphology and degradation of many stream channels associated with rheochrene springs of the Southwestern USA. Causes for the channel down-cutting of the system are attributed to anthropogenic modification of the channel with the installation of a series of low-head dams and grazing of domestic animals and introduced, non-native wildlife in the channel and the drainage basin. In cooperation with land managers, channel restoration was completed to return the function and structure of the system. Restoration included stream channel morphologic reconstruction and hydrologic and vegetative monitoring. The channel was significantly incised and the sinuosity decreased resulting in greater flow velocities, steep channel banks, and flood flows which couldn't dissipate over the flood plain.

The restoration of Hoxworth Springs included reshaping of the channel based on morphologic patterns observed in abandoned reference sections of the channel on the flood plain surface and with similar runoff dominated rheocrene spring channels in the region (Griffiths et al. 2008). Re-vegetation was performed to stabilize the restored channel banks and large exclosures were constructed to manage grazing along the channel. A three-dimensional groundwater flow model was created to help interpret and predict effects of the restoration effort on perennial stream discharge, effectiveness of the restoration, and system response to climatic extremes. The model demonstrated that the length of perennial flow in the channel was dependent on the recent climate conditions. The use of a groundwater model to evaluate restoration efforts allows the user to modify recharge conditions based upon climatic or hydrologic perturbations and estimate impacts to the length of perennial flow and water availability to the riparian ecosystem.

13.6 Concluding Remarks

We now have, for the first time, a range of tools that cover the full temporal and spatial scales across which ecology moves (seconds-to-decades; from leaf-to whole-of-catchment). Measurements of stomatal or canopy conductance, sapflow, canopy temperature, leaf area index and rates of evapotranspiration and productivity can be made using ecophysiological techniques and remote sensing technologies. These data can be used in simple, moderate and complex models of ecosystem structure and function to identify the presence, areal extent and health of GDEs. What remains to be done? The three largest knowledge gaps are, in our opinion, (1) definition of the response function of ecosystems to changes in groundwater availability or groundwater quality; (2) determination of the threshold for GDEs beyond which unacceptable changes in GDE structure and function occur; and (3) a mechanistic understanding (and hence predictive capacity) of the interaction of future climate variability on GDEs.

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Interactions of Water Quality and Integrated Groundwater Management: Examples from the United States and Europe

14

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Abstract

Groundwater is available in many parts of the world, but the quality of the water may limit its use. Contaminants can limit the use of groundwater through concerns associated with human health, aquatic health, economic costs, or

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even societal perception. Given this broad range of concerns, this chapter focuses on examples of how water quality issues influence integrated groundwater management. One example evaluates the importance of a naturally occurring contaminant Arsenic (As) for drinking water supply, one explores issues resulting from agricultural activities on the land surface and factors that influence related groundwater management, and the last examines unique issues that result from human-introduced viral pathogens for groundwater-derived drinking water vulnerability. The examples underscore how integrated groundwater management lies at the intersections of environmental characterization, engineering constraints, societal needs, and human perception of acceptable water quality. As such, water quality factors can be a key driver for societal decision making.

14.1 Introduction

Groundwater is commonly found in most parts of the world, but the quality of the water may be sufficiently poor to preclude or limit its use. Contaminants that affect groundwater use are related to human health, aquatic health, economic costs, or even societal perception. In this way, water-quality drivers might be considered different from factors of integrated groundwater management (IGM) covered in Chap. 1 and other chapters. For example, in their commentary on defining water quality, Chapelle et al. (2009) suggest the term "water quality" is inherently based on human judgments as to how water of given composition fits perceived needs, where the needs can be those of the individual, group, or ecosystem. At the same time, human judgments of water quality are dynamic. In the twentieth century water became cheap, safe, and widely available – something that had not happened before during the whole of human history (Fishman 2011). Such dynamic views can become drivers that inform current opinion and perceptions of water quality in the twenty-first century. In addition, constantly improving technology for water quality characterization identifies more contaminants at lower detection limits, which contributes to the dynamic perception of water quality, including whole new classes of contaminants (e.g., Focazio et al. 2008). How such issues are handled in a management framework can influence the subjective idea of water quality. In this way, IGM forms an important intersection of environmental characterization (e.g., water chemical analyses), engineering (e.g., water treatment and sanitation), societal needs (e.g., food supply), and human perception of water quality. This intersection of disparate drivers can, in turn, act as a key driver for societal cost-benefit analyses and other decision making.

How do we judge if water quality is limiting availability? For some contaminants and uses, objective water-quality criteria are available. For example, risk-based regulatory limits have set threshold quantities such as a "Maximum Contaminant Level (MCL)" or, a less stringent, "Preventative Action Limit" (PAL) used in the United States and similar thresholds in other countries (Table 14.1). Yet, subjective judgments can also affect perceptions of water quality, thus making acceptable water quality a dynamic interpretation.

Table 14.1 Comparison of drinking water-quality standards and guidelines for the World Health Organization, European Union, Australia, United States, and Canada. All standards and guidelines in mg/L (modified from Boyd (2006) with updates to United States as of 2013 http://water.epa.gov/action/advisories/drinking/upload/dwstandards2012.pdf)

Chemical	WHO	E.U.	Australia	U.S.	CANADA
1,1-Dichloroethylene	0.03	–	0.03	0.007	0.014
1,2-Dichlorobenzene	1	0.0001	1.5	0.6	0.2
1,2-Dichloroethane	0.03	0.003	0.003	0.005	0.005
1,4-Dichlorobenzene	0.3	0.0001	0.04	0.075	0.005
2,3,4,6-tetrachlorophenol	–	0.0001	–	–	0.1
2,4,6-trichlorophenol	0.2	0.0001	–	–	0.005
2,4-D	0.03	0.0001	0.0001	0.07	0.1
2,4-Dichlorophenol	–	0.0001	0.2	–	0.9
Aldicarb	0.01	0.0001	0.001	0.003	0.009
Aldrin/Dieldrin	0.00003	0.00003	0.00001	–	0.0007
Antimony	0.02	0.005	0.003	0.006	0.006
Arsenic	0.01	0.01	0.007	0.01	0.01
Atrazine	0.002	0.0001	0.0001	0.003	0.005
Azinphos-methyl	–	0.0001	0.002	–	0.02
Barium	0.7	–	0.7	2	1
Bendiocarb	–	0.0001	–	–	0.04
Benzene	0.01	0.001	0.001	0.005	0.005
Benzo[a]pyrene	0.0007	0.00001	0.00001	0.0002	0.00001
Boron	0.5	1	4	–	5
Bromate	0.01	0.01	0.02	0.01	0.01
Bromoxynil	–	0.0001	0.03	–	0.005
Cadmium	0.003	0.005	0.002	0.005	0.005
Carbaryl	–	0.0001	0.005	–	0.09
Carbofuran	0.007	0.0001	0.005	0.04	0.09
Carbon tetrachloride	0.004	0.0001	0.003	0.005	0.005
Chloramines-total	–	–	3	4	3
Chlorpyrifos	0.03	0.0001	0.01	–	0.09
Chromium	0.05	0.05	0.05	0.1	0.05
Cyanazine	0.0006	0.0001	–	–	0.01
Cyanide	0.07	0.05	0.08	0.2	0.2
Cyanobacterial toxins	–	–	0.0013	–	0.0015
Diazinon	–	0.0001	0.001	–	0.02
Dicamba	–	0.0001	0.1	–	0.12
Dichloromethane	0.02	–	0.004	0.005	0.05
Diclofop-methyl	–	0.0001	0.005	–	0.009
Dimethoate	0.006	0.0001	0.05	–	0.02
Dinoseb	–	0.0001	–	0.007	0.01
Diquat	–	0.0001	0.0005	0.02	0.07
Diuron	–	0.0001	0.03	–	0.15
Ethylbenzene	0.3	–	0.3	0.7	–

(continued)

Table 14.1 (continued)

Chemical	WHO	E.U.	Australia	U.S.	CANADA
Fluoride	1.5	1.5	1.5	4	1.5
Glyphosate	–	0.0001	0.01	0.7	0.28
Lead	0.01	0.01	0.01	0.015	0.01
Malathion	–	0.0001	–	–	0.19
Mercury	0.001	0.001	0.001	0.002	0.001
Methoxychlor	0.02	0.0001	0.0002	0.04	0.9
Metolachlor	0.01	0.0001	0.002	–	0.05
Metribuzin	–	0.0001	0.001	–	0.08
Monochlorobenzene	–	–	–	0.1	0.08
Nitrate	11	11	11	10	10
Nitrilotriacetic acid	0.2	–	0.2	–	0.4
Paraquat	–	0.0001	0.001	–	0.01
Parathion	–	0.0001	0.01	–	0.05
Pentachlorophenol	0.009	0.0001	–	0.001	0.06
Phorate	–	0.0001	–	–	0.002
Picloram	–	0.0001	0.3	0.5	0.19
Selenium	0.01	–	0.01	0.05	0.01
Simazine	0.002	0.0001	0.0005	0.004	0.01
Terbufos	–	0.0001	0.0005	–	0.001
Tetrachloroethylene	0.04	0.01	0.05	0.005	0.03
Toluene	0.7	–	0.8	1	–
Trichloroethylene	0.07	0.01	–	0.005	0.005
Trifluralin	0.02	0.0001	0.0001	–	0.045
Trihalomethanes	–	0.1	0.25	0.08	0.1
Uranium	0.015	–	0.02	0.03	0.02
Vinyl chloride	0.0003	0.0005	0.0003	0.002	0.002
Xylenes-total	0.5	–	0.6	10	–

This chapter will present three examples that demonstrate how water quality factors can influence groundwater use and related management options. The examples are intended to present: (1) an overview of mechanisms of how water quality affects IGM; (2) a short listing of classes of contaminants that have affected groundwater use; and (3) a description of issues and associated IGM responses that have been used to address classes of water quality issues. Because the range of potential societally-relevant water quality issues is large, we focus here on transferable elements contained within the examples. Using the dimensions of integrated groundwater management outlined in Chap. 1, water quality can be seen as integration of both natural and human systems across multiple scales of space and time. Moreover, a definition of adequate water quality is highly dependent on stakeholders, as well as new methods of identifying and quantifying contaminants. It should be noted that some water quality topics are also covered separately in more detail elsewhere in this book, including salinity (Chap. 15).

14.2 Contaminants that Affect Acceptable Water Quality Determinations

For convenience, contaminants are grouped into two broad categories that affect groundwater use: naturally occurring contaminants and human-introduced contaminants. Such a distinction cannot hold universally– for example, human activities such as high capacity pumping change the aquifer geochemical environment, which in turn can mobilize contaminants or transform them into different forms. Likewise, salinity is naturally occurring, but also can be a water quality concern in areas where it is not naturally occurring as a result of human use such as application of salt to prevent road icing. Our distinction is more robust, however, when considering the primary sources of contaminants and how they propagate to issues of water quality. Therefore, our discussion here follows this overarching criterion.

Table 14.2 lists a number of naturally occurring and human-introduced contaminants that can potentially influence groundwater management. Potential management actions to address water quality may include, but are not limited to, strategies involving:

- Source removal (e.g., centralized waste digesters, integrated pest management plans, organic farming)
- Tiered water quality designations that allow reuse of "grey water" or use of waters naturally having lesser quality (e.g., brackish groundwater)

Table 14.2 Common contaminants listed as a source of poor water quality

A. Naturally occurring contaminants
i. Salinity (Richter and Kreitler 1991; vanWeert et al. 2009)
ii. Radionuclides (Focazio et al. 2000; Szabo et al. 2012)
iii. Manganese (World Health Organization 2011)
iv. Total dissolved solids, iron, and aesthetic contaminants (DeSimone et al. 2009; Warner and Ayotte 2014)
v. Arsenic (e.g., see Sect. 14.3.1)
B. Human-introduced contaminants
a. Non-pathogen
i. Chloride (Granato 1996; Mullaney et al. 2009)
ii. PCBs/PAHs
iii. Nutrients/nitrate (Dubrovsky et al. 2010)
iv. Pesticides (e.g., see Sect. 14.3.2)
v. Non-Aqueous Phase Liquids (Mayer and Hassanizadeh 2005)
vi. Pharmaceuticals/personal care products (Barnes et al. 2008)
vii. VOCs (Zogorski et al. 2006)
b. Pathogens
i. Bacteria (Hynds et al. 2014)
ii. Viruses (e.g., see Sect. 14.3.3)

- Blending of water supplies from different sources to meet regulatory limits
- Modifying well open intervals or pumping regimes to minimize poor water quality
- Artificial aquifer recharge or aquifer storage and recovery systems
- Source minimization (e.g., landuse restrictions in wellfield capture areas, voluntary conservation)
- Water treatment at wellhead or point-of-use
- Wastewater treatment

These actions are often used in combination, and span a range of capital cost incurred during initial implementation as well as on-going cost of operation and maintenance. As might be expected given the range of cost and range of potential concerns shown in Table 14.2, there is no single or universally recommended approach for addressing water quality issues in an integrated groundwater management framework. Therefore, examples of groundwater management are used to illustrate applications where one or more of the actions described above were considered.

14.3 Three Examples of Water Quality Issues and Integrated Groundwater Management

In this Sect. 14.3 Case studies are presented here that use one naturally occurring and two human-introduced contaminants to illustrate the intersection of water quality and integrated groundwater management. Each will discuss the contaminant sources, health/aquatic/economic implications, factors affecting contaminant transport and transformation, and management solutions investigated.

14.3.1 Naturally Occurring Contaminant: Arsenic

Arsenic (As) is a contaminant that is commonly derived from natural sources and has affected the availability or use of groundwater. This case study of arsenic illustrates the importance of integrating water quality into groundwater management. People and policy makers in many parts of the world – but especially in South Asia and North China Plain–are aware of the dangers of drinking poor quality groundwater high in arsenic (Mukherjee et al 2006; Sharma et al. 2006; Singh et al 2014). Other studies predicting the occurrence of arsenic worldwide suggest that arsenic concentrations of human-health concern can be expected over large regions (Fig. 14.1) (Welch et al. 2000; Smedley et al. 2002; Amini et al. 2008; Winkel et al. 2008; Van Halem et al. 2009). Integrated groundwater management

Fig. 14.1 Arsenic affected countries (*red*) of the world (From Van Halem et al. 2009)

for arsenic is a function of: (1) understanding the spatial and vertical extent of the problem by monitoring; and (2) managing human activities, such as pumping or locating landfills, that can change the geochemical conditions of the aquifer and mobilize arsenic.

Health effects from exposure to arsenic in drinking water include increased risk for bladder, skin, kidney, and lung cancers, and increased risk for diabetes and heart disease (National Research Council 2001). Research on the health effects of low-to-moderate concentrations of arsenic caused the U.S. Environmental Protection Agency (USEPA) in 2006 to lower the MCL from 50 to 10 μg/L illustrating how new research and information can change the perception of acceptable water quality. Many countries have similar drinking water-quality guidelines for arsenic and other contaminants (Table 14.1). The United States, European Union, and World Health Organization consider 10 μg/L of arsenic acceptable for drinking water (Boyd 2006).

Integrated groundwater management can mean appreciable resources are needed for monitoring and characterizing the extent and changes in arsenic concentration. For example, in the United States testing for arsenic in publicly-supplied drinking water is part of the Safe Drinking Water Act so public supplies are monitored regularly. Yet over 43 million people in the United States get their drinking water from privately owned household wells (DeSimone 2009). The quality and safety of these privately-owned water supplies are not regulated under Federal, or in most cases state, law. Individual homeowners are responsible for maintaining their water supply systems and for any routine water-quality monitoring. The U.S. Geological Survey National Water Quality Assessment Program (NAWQA) included sampling of more than 2100 privately owned wells in the United States (DeSimone 2009) and found that about 7 % of privately owned wells contained arsenic greater than 10 μg/L. In some areas, such as the methanogenic parts of the glacial aquifer system, up to 50 % of the privately owned wells had arsenic concentrations greater than 10 μg/L (Thomas 2007). The publicly supplied drinking water is managed because routine monitoring identifies the high arsenic concentrations that need to be addressed, yet voluntary self-monitoring of privately owned wells is not routine. Identification of the problem is a first step for IGM.

Monitoring over time to assess seasonal changes in water-quality concentrations imply that there is not a one-size-fits-all solution to water-quality management over a year. A study in Albuquerque, New Mexico, shows that arsenic concentrations vary spatially and temporally in water from public-supply wells partly because groundwater with different arsenic concentrations migrates between different parts of the basin-fill aquifer within the wellbores of idle supply wells (Eberts et al. 2013). During times when the wells are not pumping, high-arsenic groundwater from deep within the aquifer moves up and out into the shallow parts of the aquifer in areas where hydraulic gradients are upward. When pumping resumes, arsenic-laden water enters these wells from both shallow and deep parts of the aquifer. Concentrations in the produced water are then elevated until the high-arsenic water is purged from the shallow parts of the aquifer. Public-supply wells in this area are pumped less frequently in the winter than in the summer so arsenic concentrations are highest in winter water samples from the deepest wells in the parts of the aquifer having upward hydraulic gradients. Well construction (depth), well operation (duration of pumping), and position within the groundwater-flow system (location with respect to vertical hydraulic gradients) affect high arsenic concentrations in water from public-supply wells. Monitoring changes in pumping and arsenic concentrations over time will enable resource managers to better manage concentrations in the produced water by pumping existing wells for longer periods during the winter and by installing new supply wells at shallower depths in certain areas (Laura Bexfield, U.S. Geological Survey, written commun., 2012).

Naturally occurring contaminants like arsenic are ubiquitous in many aquifer systems and the identification of the processes that control their mobilization and transport could help water managers meet compliance standards (e.g., Gotkowitz et al. 2004). Solid-phase chemistry data are useful in understanding arsenic sources, but do not always correspond to the relative concentrations in ground water (Brown et al. 2007). The transport of arsenic to drinking water wells is controlled by physical and geochemical processes.

Physical processes such as preferential flow paths, human induced and natural, can result in faster travel times and higher concentrations of arsenic in public-supply wells. Brown et al. (2007) identified preferential flow paths that include zones of high permeability in sand and gravel aquifers, conduit flow in karst aquifers, downward well-bore flow in a public-supply during periods of low or no pumping, and short-circuit pathways through wells and boreholes open to multiple aquifer layers. Methods using geophysical techniques, depth-dependent sampling, and sampling of monitoring wells adjacent to public supplies, improve the understanding of preferential flow paths and other factors such as redox chemistry and competing ions that affect the movement of arsenic to public-supply wells.

Groundwater age information is a tool that adds to our understanding of the processes resulting in elevated arsenic. For example, in the glacial aquifer system, arsenic concentrations above the drinking water standard (10 micrograms per liter (μg/L)) were most often associated with groundwater that recharged the aquifer system prior to the 1950s. Similarly, Eberts et al. (2013) found arsenic concentrations in water from public-supply wells in study areas in California, Connecticut, Ohio, Nebraska, Nevada, and Utah increased with increasing travel times to the wells (increasing groundwater age). The groundwater-age mixture for a well characterizes the complete range of time that it might take contaminants that are released to the groundwater to reach a well. An estimate for the groundwater-age mixture for a well is a useful measure of the potential for elevated arsenic in water from the well. In addition, public-supply well construction and operation (screen placement, pumping rates and schedules) can lead to differences in the age mixture of the groundwater pumped from different wells, including wells within the same aquifer. Many of the public supplies sampled as part of the NAWQA study showed a mixture of groundwater ages. This indicates that groundwater management practices need to consider natural and human-induced changes in the aquifer geochemistry over time.

Mixing of groundwater from different parts of the aquifer system can change the chemistry of the groundwater and the potential for elevated arsenic. Ayotte et al. (2011) show that pumping-induced hydraulic gradient changes and artificial connection of aquifers by well screens can mix chemically distinct groundwater. Chemical reactions between these mixed groundwater and solid aquifer materials can result in the mobilization of arsenic, with subsequent transport to water-supply wells. For example, near Tampa, Florida, much of the downward movement of groundwater is along flow pathways that follow natural conduits in the limestone bedrock (Jagucki et al. 2009). High-volume pumping from the wells in this study pulled shallow, oxic and low-pH water, which is capable of dissolving arsenic-bearing minerals, into deeper, anoxic and high-pH parts of the aquifer system where arsenic can remain in solution. This accelerated mixing of dissimilar waters both mobilizes arsenic from the rocks and allows it to remain dissolved in the newly mixed water.

In many areas, dissolved oxygen is an easily determined concentration that indicates the likelihood of elevated arsenic in the water. In the glacial aquifer system, United States, geochemical conditions identified by presence or absence of dissolved oxygen (less than or greater than 0.5 mg/L) is a good indicator of the likelihood of detecting (or not detecting) arsenic concentrations greater than the drinking-water standard (10 μg/L) (Warner and Ayotte 2014). Human activities can alter recharge or change groundwater flow in ways that lead to changes in the aquifer geochemical conditions (Eberts et al. 2013). These changes result in chemical reactions between the groundwater and the solid aquifer material, releasing naturally occurring arsenic into the groundwater. As a result, concentrations of arsenic in water from wells increases. Similarly, Gotkowitz et al. (2004) found that drawdowns resulting from pumping created conditions that mobilized naturally occurring mineralized arsenic quickly in drinking water wells that historically were not characterized as having arsenic contamination.

Other human activities can cause local and regional scale changes in aquifer geochemical conditions and indirectly increase arsenic concentrations in groundwater and in water from public-supply wells. For example, groundwater in the vicinity of a landfill can have elevated concentrations of arsenic, yet the source of the arsenic is not the contents of the landfill (Warner and Ayotte 2014). Rather the source is geologic – part of the solid aquifer material (Delemos et al. 2006). This type of situation occurs because microorganisms degrade large amounts of organic carbon derived from the waste within the landfills, creating anoxic conditions in the groundwater. Arsenic is then released from the solid aquifer material to the groundwater under the newly anoxic conditions, thus increasing arsenic concentrations in groundwater downgradient from the landfill.

Water managers who understand how redox conditions are distributed within an aquifer system are in a position to anticipate which chemical constituents in the groundwater (for example, nitrate, arsenic, iron, manganese, and certain VOCs or pesticides) would (or would not) be expected to occur in water from a particular well. In addition, knowledge about redox conditions in an aquifer system can help water managers select the most suitable water-treatment methods for water from their wells. Redox conditions of groundwater also are important because the oxidation state of some elements affects their toxicity. For example, the oxidized form of chromium (hexavalent chromium, Cr6+) is more toxic than the reduced form (trivalent chromium, Cr3+) (Mills and Cobb 2015). Another way that human activities can affect concentrations of natural contaminants in groundwater is by altering groundwater flow so that waters with different chemical characteristics mix.

Human-induced alteration of groundwater flow patterns can affect concentrations of naturally occurring trace elements like arsenic. Adverse water-quality impacts attributed to human activities are commonly assumed to be related solely to the release of the various anthropogenic contaminants at the land surface; yet, human activities including various land uses, well drilling, and pumping rates and volumes can adversely impact the quality of water in supply wells indirectly, when associated with naturally-occurring trace elements in aquifer materials (Ayotte et al. 2011). This occurs because subtle but significant changes in geochemistry are associated trace element mobilization as well as enhancing advective transport processes.

Sources of natural contaminants like arsenic are largely distributed and not usually mitigated with source remediation. The cost of treating for arsenic in large public-water utilities is an economic cost, but the human health cost of not treating for elevated arsenic in drinking water can be substantial. Costs, like that of public water suppliers using the glacial aquifer system in the United States, were estimated at 29 million dollars in 1999 to treat groundwater for a single issue of concern: elevated arsenic concentrations (Warner and Ayotte 2014). In the United States in 2006 when the drinking water standard was lowered to 10 μg/L the Illinois Environmental Protection Agency estimated that the initial cost to reduce arsenic concentrations to below the MCL of 10 μg/L for 50 of the community water supplies with elevated arsenic concentrations in Illinois (Fig. 14.2) could reach a total of $40 million dollars, with the highest costs associated with small community supplies (Warner and Ayotte 2014; Warner et al. 2003; Warner 2001). On a national or worldwide scale, this is a large water-quality cost to consider. Understanding the

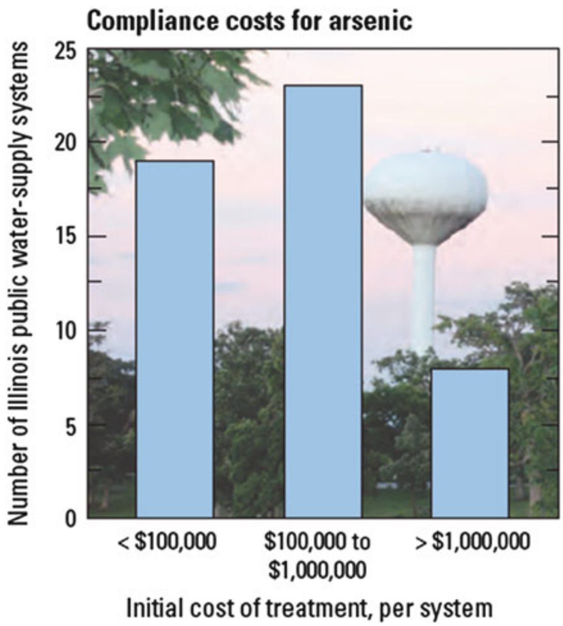

Fig. 14.2 Cost of treating drinking water for elevated arsenic (From Warner and Ayotte 2014)

processes that affect the mobilization of arsenic in groundwater leads to more informed and integrated water management decisions in areas where arsenic is a concern, which in turn can provide cost savings.

14.3.2 Human-Introduced Contaminant (Abiotic): Agricultural Inputs

The pervasive use of organic and inorganic chemicals in agricultural areas has led to the deterioration of the quality of groundwater and surface water, and has become a concern for human consumption over the last decades. Water quality deterioration by pesticides, for example, is well recognized, for surface or drained water (Schiavon and Jacquin 1973; White et al. 1967) and groundwater (Muir and Baker 1978). Since the early identification of the concern, degradation of water quality by pesticides become widespread in Europe (Capriel et al. 1985; Heydel et al 1999; Réal et al. 2001, 2004; European Commission 2002, 2010). Many recent studies have reported the presence of pesticides higher than the European regulatory limits of 0.1 μg/L and 0.5 μg/L for surface water and groundwater, respectively. In one survey, total concentration of pesticides was over 0.5 μg/L in 18 % of surface water samples and 3.8 % of groundwater samples analyzed (SOeS 2010).

With the expected conflicting goals of crop production and preservation of surface and groundwater quality, an integrated water resources management approach is needed. Integrated groundwater management, specifically, must embrace spatial and temporal uncertainty both in the source (due to changing human application rates and chemical properties) and in the groundwater aquifers that embody a heterogeneous application and transport of that source. Even defining the groundwater system of interest can be problematic because: 1) groundwatersheds can be difficult to delineate accurately and often do not align with the easily delineated overlying surface watershed (e.g., Hunt et al. 1998; Winter et al. 2003); 2) the amount of effort expended on the characterization is likely not equal in space and time in an area of interest; and 3) the land surface encompasses different political boundaries, which may change the regulatory agency charged with the management of the water resource. Integrated groundwater management must also address the fact that a groundwater system is buffered by an unsaturated zone that separates the land surface where pesticides are applied from the aquifer used. This buffering can affect the timing and amount of recharge to the water table – effects that change as the unsaturated zone thickness changes (e.g., Hunt et al. 2008). Delays and lags between an activity, or change in activity like Best Management Practices, at the land surface and its appearance in the groundwater resource can confound simple cause-and-effect relations that underpin decision making.

For agricultural contaminants, integrated groundwater management is a function of: (i) changes in protective areas specified at land surface that can determine and influence the contaminant source; and (ii) the importance of lags and delays between the driving forces at the land surface and the change of the groundwater resource.

14.3.2.1 Changing Protective Areas at the Land Surface

Here we use two French groundwater systems as examples, the Vittel and Lons-le-Saunier catchments located near the French-Swiss border. Vittel watershed is managed through voluntary agreements between diverse stakeholders and the private enterprise Nestlé Water (Benoît et al. 1997). The Vittel catchment has been the focus of a delineation process since 1925 (Barbier and Chia 2001). The catchment outline defined during negotiations with farmers and other stakeholders began in 1987 was 4200 ha. In 1994, new hydrological work increased the catchment to 4500 ha. In the case of Lons-Le-Saunier, the catchment is managed by the municipality and a group of priority catchment organizations; they are called the "Grenelle Catchments" because they were designated through the Grenelle Initiative – a collection of political meetings that occurred during the fall of 2007 to make long-term decisions on sustainable development. The Lons-le-Saunier catchment also will likely have multiple delineations (Hellec et al. 2013; Barataud et al. 2014a).

Areas identified for protection within the delineated groundwater resource have also evolved over time as a result of increasing awareness of contamination, negotiations with the farmers, and the evolution of the driving regulatory context (from Public Health Laws to a patrimonial management of water in the recent Environment Code). Today the management zone is divided into four zones (Fig. 14.3). The water wells zone (zone I) of about 7 ha, without any agricultural activity, was bought by the municipality in 1961 at the beginning of the wells' use.

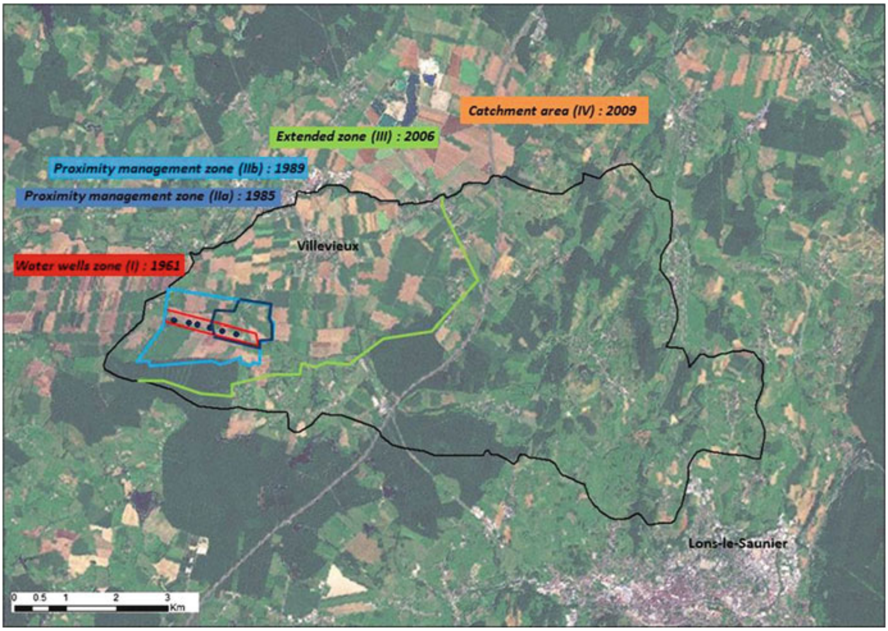

Fig. 14.3 Example of successive delimitation of the protection perimeters

A proximity management zone with two sub-divisions was then defined: contracts between the municipality and the farmers were primarily established on a zone IIa (63 ha) in 1985 when nitrates and atrazine were noted in the wells; zone IIa was extended to a zone IIb (220 ha) in 1989 and the contracts were re-negotiated in 2006 as a new French regulatory requirement imposed a more formalized definition of protection perimeters. In 2006 the zone was again extended to include an additional 1500 ha (zone III). Currently, the protection zones consist of slightly less than 1800 ha, corresponding to about 30 % of the total catchment area. The total catchment was designated in 2009 as zone IV, defined using the hydrological report that resulted from the 2009 Grenelle Initiative.

Concurrently, a 1992 French law of the Public Health Code required a mandatory "Declaration of Public Utility" for water resources, which included a delineation of water protection areas in which conservation easements can restrict agricultural practices. In practice, the delineation of public utility is commonly delayed. A recent study showed that only two-thirds of catchments in the French Grenelle priority catchment were in conformity for the delineation of water protection areas (Barataud et al. 2014b), whereas a deadline of 5 years was given by the 1992 law . Local stakeholders noted a high level of inter-stakeholder conflict caused by these regulatory requirements. Using the catchments that are in conformity with the 1992 law, it is clear there is a wide range of management unit size (Fig. 14.4).

The size of the management unit can affect execution of protective measures. Perhaps most obviously, developing mutually agreeable solutions with the

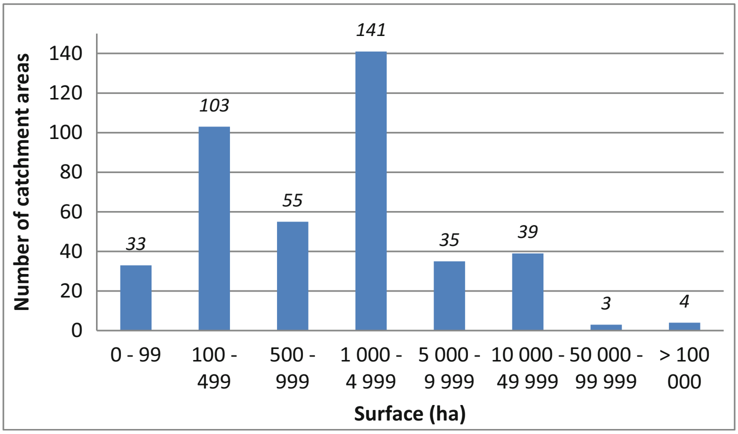

Fig. 14.4 Variability of the protection area size within the priority Grenelle catchment

agricultural producers and other stakeholders in large catchment areas is more difficult because there are more entities to include, and is often hindered by simple organization challenges such as identifying meeting-times and discussion frameworks. In large catchments, accounting for the interests and wide ranging viewpoints often requires designation of intermediaries to facilitate discussion that represent the whole of the stakeholder group. In small catchment areas, protective practices may be identified but often involve improved agricultural practices over only small parts of the catchment rather than major farming practice reforms. Several studies have questioned the effectiveness of partial measures for protecting and restoring target groundwater resources (Kunkel et al. 2010; Thieu et al. 2010; Lam et al. 2011; Posen et al. 2011).

14.3.2.2 Temporal Characteristics of Groundwater Management

Clearly the spatial area included or excluded from a protective action will influence the associated groundwater quality. Temporal aspects can also affect integrated groundwater management. The temporal aspects covered here include timing of human implementation of protective measures at the land surface, and time lags that result from the natural groundwater system itself.

An example of the human dimension is seen in the 2000 European Water Framework Directive (WFD), which proposed three new articles: preservation of water bodies as a whole (taking into account non-point pollution and not just point-source pollution), an imposed schedule for adoption, and objectives defining quantified results for ecological restoration of the environment. This Directive is complex and ambitious, but is considered a cornerstone of the European Union's environmental policy (Bouleau and Richard 2009). France partially conformed to this directive 6 years after the Directive was signed through its Law on Water and Aquatic Environments (Loi sur l'Eau et les Milieux Aquatiques [LEMA], 2006), where for the first time in French law the definition of non-point pollution appeared.

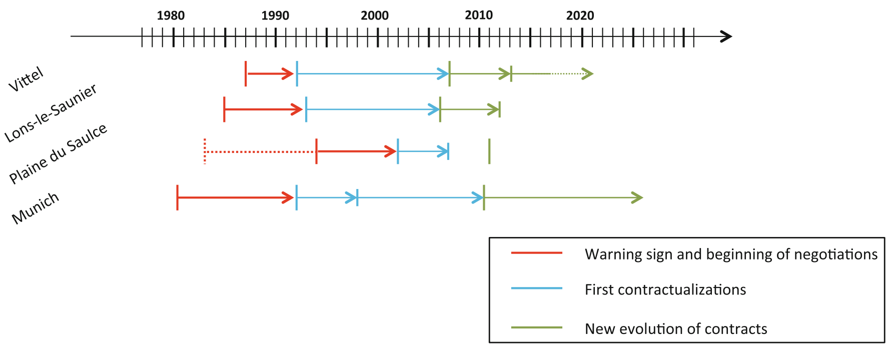

Fig. 14.5 Timelines of protection activities of catchments in four areas

However, it was not until the Grenelle Initiative in 2009 and the designation of the Grenelle priority catchments, that the notion of schedules, deadlines, and quantifiable results was written into French law. In the example of Lons-le-Saunier, 9 years were necessary to partially translate the WFD into application in one area of France, and the process was considered difficult by most all involved.

The human dimension also can result in unintended parallel protective actions. Faced with insufficient regulatory frameworks, many local water managers (municipalities, water utilities, private entities) outside of the Grenelle priority catchments have set up, or are currently setting up, their own coordination with farmers to promote protective practices to enhance local water resources quality. Each protective practice imposes various time frames for adoption, many of them distant into the future, as can be seen by comparing the timelines for the above mentioned Lons-le-Saunier and Vittel Catchments to two other European catchments (Fig. 14.5: La plaine du Saulce in western France and one near Munich, Germany). The Munich catchment is notable because it is an early example of protection of water quality internally developed after adoption of organic farming practices at a near catchment scale. The time from the identification of the problem and subsequent negotiations to formal protective measures can range between 5 and 20 years. Clearly lags in the adoption of protective measures will result in lags in obtaining the improved water quality that initially drove the adoption of protective measures.

Given the competing interests of the multiple stakeholders, problem scoping activities and protective action negotiations often require many months of discussion. For example, the mobilization of stakeholders, identification of needs and priorities, negotiations between stakeholders having conflicting interests, defining a consensus, and constructing adequate institutional forms, are all necessary stages which require different amounts of time and effort to execute. Even after protective measures are adopted, it is not uncommon to see delays of several years needed to coordinate and modify individual practices.

This temporal and spatial complexity of adopted protective measures then must then filter through the natural system to where the groundwater resource of interest is assessed. Nitrate pollution management in the Plaine du Saulce catchment discussed below exemplifies how the natural system dimensions can delay positive responses in the groundwater resources resulting from management intervention to reduce contamination. The water catchment area (86 km^2) is situated 10 km south of the City of Auxerre, on a rural agricultural landscape consisting of 45 farms (4026 ha). In the early 1990s, high levels in nitrate concentration were recorded in the Auxerre groundwater wells in the early 1990s supplying one third of the 60000 inhabitants' water requirements. In 1994, peaks reaching 70 mgNO_3^-/L (exceeding the European drinking standard of 50 mgNO3-/l) precipitated a lively debate on management strategies to deal with this nitrate contamination. Various managing entities were brought to bear over the next three decades, with the first contract with farmers in 2002, 8 years after the first sign of severe degradation. The management strategy initially operating on a voluntary basis did not result in significant decrease in the nitrate concentrations. As a result, regulation was proposed in 2011 focusing on integrated agriculture, where adoption would become a mandatory after a period of 3 years. The proposed regulatory framework caused major tensions between stakeholders, made worse by a lack of understanding regarding the absence of improved water quality after many years of joint protective actions.

During 2012, a scientific committee met twice to update management strategies to account for the natural delay between changes in agricultural practices at the land surface and measurable improvements in water quality. One primary conclusion was that groundwater flow rates in the Sequanian limestone aquifer tapped by the wells are relatively longer than human timeframes considered in management actions. Water dating analysis through anthropogenic tracers CFC and SF_6 estimated an aquifer residence time of around 25 years (±3 years) at the pumping wells (Anglade et al. 2013). As a result, nitrate levels observed at the wells reflected agricultural practices that occurred over two decades ago. Analysis of agricultural nitrate use also supported this assessment. Nitrogen inputs had sharply increased in the 1960s before stabilizing in the 1990s (Fig. 14.6); point-to-point comparison between nitrogen surplus and measured nitrate concentration also suggested an approximately 25 year lag in response at the wells.

This example underlines that when planning and implementing management actions, expected time lags need to be communicated to stakeholders and funding agencies in order to reduce short-term expectations that may impair long-term political and financial support. At this point in the Plaine du Saulce catchment, such knowledge has opened up new possibilities for organic farming, with recognition that changes are needed beyond the catchment borders.

Human-introduced pesticides also represent challenges to integrated groundwater management. They can affect the quality of drinking water; especially groundwater close to land surface (e.g., Schreiber et al. 1993). Many pesticides can persist for long periods in the environment – organochlorine insecticides, for example, were still detectable in surface waters 20 years after their use had been banned (Larson et al. 1997). Others studies documented measurable pesticide concentrations years after their last application on the land surface (Baran

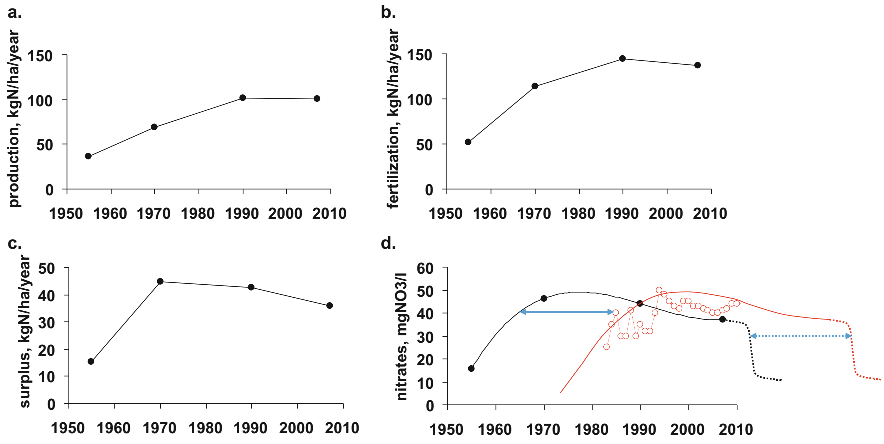

Fig. 14.6 (**a**) and (**b**) Evolution of harvested nitrogen and total nitrogen inputs (synthetics and organic fertilizers, atmospheric dry and wet deposition, biological nitrogen fixation) on arable land since 1950. (**c**) Calculation of N surplus (Harvested N – Total N inputs). (**d**) Resulting nitrates concentration (infiltration flux of 240 mm/year) and comparison with recorded nitrates levels in the wells (*red points*)

et al. 2008; Buhler et al. 1993; Jarczyk 1987; Novak et al. 1998; Reiml et al. 1989). In France, Atrazine was banned in 2003; yet, analysis of the Grenelle priority catchment area suggests that half of the protected catchments have measurable atrazine or atrazine degradation product, called a metabolite, in 2011 (Barataud et al. 2014b).

Site-scale studies have been used to help explain the persistence of pesticides in groundwater (Perrin-Ganier et al. 1996). In one case study by Schrack et al. (2009, 2012) from the Lorraine region of France, agricultural practices had been recorded annually for 40 years, including pesticides use during conventional crop management (date, product, application rate). From September 2004 to the present, no pesticides have been used on the study fields as a result of conversion to organic farming practices. During the 30-year period prior to conversion to organic practices, many pesticides were applied on crops, including herbicides atrazine and 2,4-D (2,4-dichlorophenoxy acetic acid). Similar to the observations of Barataud et al. (2014b), measurable atrazine was documented over 10 years after atrazine application ceased. 2,4-D concentrations were higher than the regulatory limits in two water samples from drain tiles (Fig. 14.7), despite low detection frequency in point samples at the site. Thus it appears that even though the soil zone can reduce and transform pesticides applied to crops, it can also act as a diffuse source of groundwater contamination that persists after application ceases. That is, organic farming initiated in 2004 does not apply pesticides; however, more than 5 years after conversion to organic farming practices, pesticide concentration can still exceed the regulatory limit (e.g., 2,4-D drain water in Fig. 14.7).

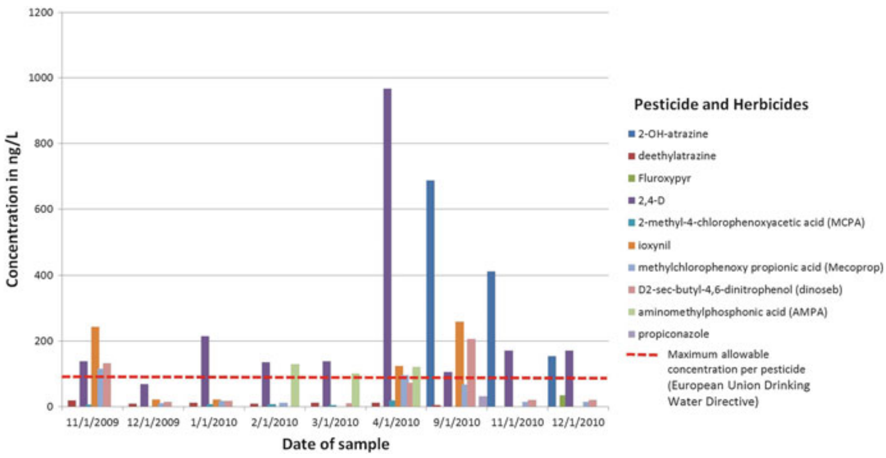

Fig. 14.7 Concentration of pesticides in experimental field after stopping their spreading on the experimental field (2,4-D: since 17 years; Ioxynil: since 13 years; Mecoprop: since 21 years; Dinosèbe: since 15 years; Atrazine: since 23 years; DEA: since 16 years; AMPA-glyphosate: since 17 years)

14.3.3 Human-Introduced Contaminant (Biological): Human Enteric Viruses

As shown in Tables 14.1 and 14.2 many types of human-source contaminants can influence groundwater management, and make an otherwise acceptable groundwater supply not suitable for an intended use. Agricultural contaminants, presented in Sect. 14.3.2 are a widely recognized example. Here we discuss a less known human contaminant – human enteric viruses, a subset of possible biological entities, called pathogens, that can affect drinking water suitability. Although the importance of viruses as a groundwater contaminant is primarily restricted to human drinking water, this example helps illustrate how recent advances in methodologies for detection and quantification provide new insights into vulnerability of groundwater supplies not provided by the traditional understanding of water quality contaminants. The material in this section is taken from Borchardt et al. (2004, 2012) and Hunt et al. (2005, 2010, 2014); the interested reader is directed there for additional information.

Viruses are infectious particles of nucleic acid wrapped in protein and sometimes an outer layer of lipid that replicate only within cells of living hosts. In the environment they are metabolically inert. Virus spread is facilitated by concentrated sources and the very low exposure needed for infection. For example, a gram of feces from an infected host can contain trillions of infectious viruses, yet only 1–10 viruses are required to infect a new host. The human health implications of waterborne virus contamination are multi-fold. Recent studies have demonstrated occurrence of human enteric viruses in domestic and municipal wells in the United States (Abbaszadegan et al. 2003; Borchardt et al. 2003; Fout et al. 2003; USEPA 2006). Of the 248 recorded drinking water outbreaks caused by untreated groundwater in the United States between 1971 and 2008, 32 (12.9 %) had a viral etiology. Moreover, in 135 outbreaks (54.4 %) the etiology was unidentified (Wallender et al. 2013), but believed to be viral as in the early years of outbreak surveillance the technology to detect waterborne viruses was less widely available than it is today. Outbreaks related to virus-contaminated groundwater have also been documented in other parts of the world (Gallay et al. 2006; Beller et al. 1997), suggesting widespread hydrologic conditions suitable for virus survival and transport.

Viruses are much smaller (27–75 nm) than bacterial and protozoan pathogens and thus are more easily transported through pores that physically filter larger pathogens. Virus adsorption onto sediment grains is a primary removal mechanism, although the strength of adsorptive forces depends on sediment and water chemistries (Borchardt 2006). These factors notwithstanding, viruses may still be transported some distance, even into confined aquifers at travel rates relevant for human-health concern (e.g., Borchardt et al 2007; Bradbury et al. 2013). As a result, the United States Environmental Protection Agency has listed several viruses on the third drinking water Contaminant Candidate List, emphasizing that waterborne viruses are a research priority (http://www.epa.gov/ogwdw000/ccl/ccl3.html).

There is also significant public and regulatory interest in understanding the vulnerability of water-supply wells to contamination by human enteric viruses (e.g., http://www.epa.gov/safewater/ccl/index.html; Unregulated Contaminant Monitoring Rule 3 – USEPA 2011). However, assessing well vulnerability to infectious pathogens is different because pathogen vulnerability assessments require knowledge of very fast (<3 year) times of travel – a timeframe not characterized by common groundwater age dating methods (Hunt et al. 2005, 2014). Therefore, a different conceptualization is needed to assess well vulnerability to pathogens.

Plume center-of-mass approaches of contaminant transport typically define risk from non-pathogen contaminants such as those listed in Tables 14.1 and 14.2; they reflect the bulk properties of the aquifer which control transport to a drinking well where risk is calculated using long-term exposure relevant for slowly moving plumes. Pathogen transport to groundwater-supply wells is different because adverse health effects can only occur while a pathogen is still infectious; viruses are reported to remain infectious in groundwater for time periods less than 3 years (Seitz et al. 2011). However, unlike dissolved contaminants, as particles pathogens tend to follow fast preferential flow pathways with minimal matrix diffusion (McKay et al. 1993; DeBorde et al. 1999). Thus, rather than well vulnerability assessment based on decade-scale water movement, it is the fast pathway properties of the aquifer that are most important for understanding the vulnerability to pathogens and the risk for disease transmission.

For many groundwater systems, a 1–3 year travel time might be considered of little importance because distances traveled in many unstressed groundwater systems in even 3 years are short. But this is not true for all groundwater systems; large distances can be traveled in short timeframes in karst and fractured rock aquifers (e.g., Borchardt et al. 2011). Even in porous media aquifers, high capacity water-supply wells significantly depressurize local groundwater systems and create large hydraulic gradients. These gradients, in turn, result in faster local groundwater velocities than occur in natural groundwater flow systems. This could explain, in part, why virus contamination frequency tends to be greater in high capacity wells than in private domestic wells (Borchardt et al. 2003). More surprising, in the confined aquifer supplying drinking water to Madison, Wisconsin USA, there are pathways sufficiently fast that virus transport to deep supply wells cased through the aquitard can occur in several weeks (Bradbury et al. 2013).

Viruses can only be a contaminant of concern, however, if there is an infectious human fecal source. One common source is leaking sanitary sewers (Hunt et al. 2010). Reported estimates of sanitary sewer leakage, or “exfiltration”, range from 1 % to 56 % of the dry weather flow (Rutsch et al. 2008). In the United States, exfiltration has been estimated as 30 % of system flow as a result of infrastructure deterioration, and in local areas, sanitary sewer leakage has been reported to be as high as 50 % of the system flow (USEPA 1989). The exfiltration rate for a European sanitary sewer has been reported on the order of 1 l/m of sewer line per day (Lerner and Halliday 1994). Exfiltrated volumes for large municipalities are thought to reach tens of thousands of cubic meters per day (millions of gallons per day), exceeding the capacity of the sediments to filter, absorb, and immobilize contaminants carried therein (Amick and Burgess 2000). Even though more research is needed to make general system predictions (Rutsch et al. 2008; Tafuri and Selvakumar 2002), local sanitary sewers have been related to drinking-water associated outbreaks of gastroenteritis (e.g., see Amick and Burgess 2000; Bishop et al. 1998). Older, non-maintained systems are thought to be more susceptible to exfiltration, as well as systems including pressurized by sewage lift stations (Decker 1994a, b). For example, of the wells sampled by Borchardt et al. (2004), the highest number of positive virus samples was obtained from a drinking water well near a pressurized lift station. When the water table is below the utility infrastructure, exfiltrated sewage is often concentrated and transported in the trenches surrounding sanitary sewers, especially during conditions of rainfall-induced infiltration, such that they can threaten drinking-water supplies (Tafuri and Selvakumar 2002). Sanitary sewer infrastructure is often located near municipal wellheads, and carries a high viral load during periods of infections in a community (e.g., Sedmak et al. 2003; Bradbury et al. 2013). From an IGM perspective, this presents management action options: a groundwater-supplied municipality could work to minimize sewer contamination of its urban aquifer by integrating its management teams for wastewater and drinking water, making sure both teams are aware of each other’s activities that might affect the aquifer.

From a contaminant monitoring perspective, total coliform bacteria and *E. coli* – standard microbiological indicators of water sanitary quality – are rarely correlated with viruses (Wu et al. 2011), likely due to their differences in transport/ filtering and survival characteristics in an aquifer. Even with direct analysis, virus occurrence is commonly temporally sporadic when viruses are analyzed at the wellhead. Therefore, assessing drinking well vulnerability can involve a multiple samplings, perhaps more than might be used for traditional contaminant vulnerability assessments. Fortunately, water samples for viruses can now be collected inexpensively and routinely (Lambertini et al. 2008; Gibbons et al 2010; Mull and Hill 2012), which allows affordable collecting of larger sample numbers. In the early 2000s, results from viral analysis by conventional polymerase chain reaction (PCR) usually included only virus identification and presence/absence; virus quantification could only be accomplished by culture methods and these are laborious, expensive, and restricted to only a few virus groups. Now, with the advancement of real-time quantitative PCR (qPCR), the quantities of many virus types can be reliably measured with high-throughput, low cost, and less labor. Detailed genetic information on virus subtypes can also be obtained with high-throughput sequencers widely available. Therefore, from a practical standpoint, this newly developed technology has created a capability to assess well vulnerability that was not available to groundwater managers even 15 years ago.

These available technologies have also allowed the advent of a new concept in groundwater management, using viruses as tracers of young-age groundwater (Hunt et al. 2014). Because the maximum survival time for viruses in groundwater is approximately 3 years, a positive virus signal in mixed-age groundwater, in effect, zeros-out the contribution of older water and indicates young water must be present. Moreover, because different virus types infect and then disappear from the host population over time as the number of susceptible and resistant hosts changes, this creates a time-varying signal that can be tracked in the environment. When fecal waste from an infected population is released to the environment, whether from people, livestock or wildlife, the combination of virus identities and quantities in the waste becomes a "virus snapshot" for a specific point in time. Measuring this "snapshot" at suspected virus sources and waiting for it to appear at "downstream" receptors, such as a supply well, can be used to make inferences about time-of-travel to the well; wells with very young water are typically considered more susceptible to all water quality contaminants. Unlike traditional well vulnerability assessments that are relevant for contaminants carried by "high-yield slow-pathways" in the aquifer to the well, viruses as tracers for well vulnerability assessment gives information on the less-studied leading edge and early arrival of a pathogen contaminant, which is driven by preferential flowpaths that provide "low-yield fast-pathways" to the well (Hunt et al. 2010).

In areas where groundwater supplies for drinking water are not disinfected, the economic cost of virus contamination can be considerable. In an epidemiological study of 14 groundwater-supplied communities in Wisconsin that did not practice disinfection, Borchardt et al. (2012) determined that 6 % to 22 % of the acute gastrointestinal illnesses (AGI) in these communities resulted from their virus-contaminated drinking water. The economic cost of these groundwater-borne illnesses can be roughly estimated from US data on healthcare utilization and costs for AGI in young children (Cortes et al. 2009) and extending the assumption these data apply to the rest of the population. Such an assumption is likely justified for American adults 18–54 years old because in this age group the prevalence and severity of gastrointestinal illness is not much lower than that for young children

(Jones et al. 2006). From Cortes et al. (2009), for children less than 5 years old the national hospitalization rate for AGI is 0.5 %, the emergency room visit rate is 1.8 %, and the outpatient visit rate is 13.3 %. The United States median payments for AGI treatment by hospitalization, ER visit, and outpatient is $3135, $332, and $90 (reported in 2009 USD), respectively. The number of people drinking non-disinfected municipal groundwater in Wisconsin is about 100,000. If the baseline AGI rate in Wisconsin is 1 episode/person-year, about the national average, and using the midpoint of 14 % of AGI attributable to virus-contaminated groundwater, the healthcare costs in Wisconsin are approximately $500,000 USD per year. This only includes direct payment to healthcare providers. It does not include costs to the economy from work lost either by the ill person or their caregiver, nor does it include the cost of death. It also does not consider the most disease-vulnerable populations, the immunocompromised and elderly. Moreover, this estimate can be considered conservatively low because it does not account for the legal, social, and economic costs if virus-contaminated groundwater resulted in a disease outbreak. The AGI reported in the study by Borchardt et al (2012) only measured sporadic non-outbreak illnesses.

Studies by Borchardt et al. (2012) and Lambertini et al. (2011, 2012) were part of a large United States government funded epidemiological study (the Wisconsin Water And Health Trial for Enteric Risks, or WAHTER Study), designed to measure the level of illness in communities that rely on non-disinfected groundwater as their source for drinking water. Concurrent with the study, the Wisconsin Department of Natural Resources (DNR), the state agency ceded the authority for regulation of drinking water quality, was preparing to implement the United States Federal Groundwater Rule. As it became clear the 14 Wisconsin communities enrolled in the WAHTER Study had significant virus contamination of their groundwater supplies, the DNR decided to incorporate into their statewide implementation plan a change to the State drinking water code to require disinfection for all groundwater-source municipal drinking water systems in the state. The code change was approved by the DNR oversight board. However, after a statewide election in 2010, the State legislature reversed the DNR's decision and passed a bill prohibiting the DNR from requiring drinking water disinfection (http://docs.legis.wisconsin.gov/2011/proposals/ab23, accessed August 12, 2014). The bill was signed into law in 2011 (http://docs.legis.wisconsin.gov/2011/related/acts/19, accessed August 12, 2014). This statewide action was taken despite expert testimony describing the WAHTER study results and associated estimated costs to its citizens.

In an IGM context, there were factors associated with human enteric viruses that may have influenced the decision making process. A new contaminant, viruses, and a new technology, qPCR, were unfamiliar to many drinking water utilities and policymakers. People viewed the traditional pathogen indicators total coliform and *E. coli* tests as the "gold standard" for sanitary quality; if these traditional indicators were negative the water was considered acceptable. Positive tests for traditional indicators, when they occurred, were interpreted as a distribution system problem not a quality problem associated with the groundwater source itself. Such assumptions were deemed reasonable because non-disinfecting communities were not required by State code to collect microbiological samples from their drinking-water production wells, and a common perception is that the groundwater must be clean because it is filtered by soil and aquifer material, and thus can be considered microbiologically pure. Waterborne disease may have also been viewed as being events that only occurred as disease outbreaks as reported by news headlines; the concept of low-level, but measurable, sporadic disease transmission was unfamiliar. Lastly, the actions were consistent with a public view that State government should not supersede local control of drinking water regulation. A second independent study has since corroborated the WAHTER Study findings and showed heavy precipitation events result in more children seeking medical treatment for AGI in groundwater-supplied communities in Wisconsin that do not practice disinfection compared to those communities that do (Uejio et al. 2014). This study prompted a bill to reverse the disinfection prohibition but it did not move forward (Wisconsin Assembly Bill 545, https://docs.legis.wisconsin.gov/2013/proposals/ab545, accessed August 12, 2014).

14.4 Implications for IGM

Groundwater is under increasing threat from over-development, over-extraction and pollution, due to increasing population pressure, increasing living standards, industrialization, and a lack of proper management to match the demands and use patterns (see Chap. 2 for more detail). This is a global trend, although there are regional differences. The availability of groundwater with adequate quality to meet ecological and human health needs is often in direct and immediate conflict with strategies of livelihood. Competing demands for quantity and quality of groundwater can be result in fragmented management policies. These competing needs present a problem for researchers and managers to communicate the complexity of groundwater-quality changes with changing demands and uses. There is a strong need to close the gap between the perceptions of groundwater quality and understanding.

The latest technologies and approaches in groundwater modeling, laboratory analytical methods, engineering design, and economic modeling can all inform decision-making in an IGM framework, but societal subjective perceptions of water quality and societal behavior can be equally important in some circumstances. In the context of an IGM framework, water quality issues can require regulators to devote appreciable resources to managing societal perceptions and societal behavior – additional resources beyond that needed to perform the more easily recognized components of IGM such as monitoring, engineering, and risk assessment. Moreover, additional dimensions of acceptable water quality can appear as new technology becomes available, which in turn can become important forcing functions on IGM activities. In addition to changing technology, increasing the sampling frequency used in traditional groundwater monitoring assessments can influence IGM activities. For example, infrequent sampling (often once a year) and long-term exposure risk assessment approaches may not adequately represent the dynamism of the groundwater system quality – for either pathogen or non-pathogen concerns (e.g., Hunt et al. 2010). New advances in monitoring continuous water quality, such as specific conductance and other parameters, show that changes can occur within hours or days of a precipitation event depending on the system. On the other hand, the time lag between actions at the land surface and expression in the groundwater system must also be accounted. Clearly identifying and characterizing potential water quality drivers is the first step for a successful IGM framework. From such an understanding associated risks can be estimated, which in turn can form the basis of societal discussion of costs and benefits that will form the foundation for all IGM activities that follow.

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Soil and Aquifer Salinization: Toward an Integrated Approach for Salinity Management of Groundwater

15

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Abstract

Degradation of the quality of groundwater due to salinization processes is one of the key issues limiting the global dependence on groundwater in aquifers. As the salinization of shallow aquifers is closely related to root-zone salinization, the two must be considered together. This chapter initially describes the physical and chemical processes causing salinization of the root-zone and shallow aquifers, highlighting the dynamics of these processes and how they can be influenced by irrigation and drainage practices, thus illustrating the connectivity between soil and groundwater salinization. The processes leading to aquifer salinization in both inland and coastal areas are discussed. The roles of extractive resource industries, such as mining and coal bed methane operations, in causing aquifer salinization are also outlined. Hydrogeochemical changes occurring during salinization of aquifers are examined with the aid of Piper and Mixing Diagrams. The chapter then illustrates the extent of the problem of groundwater salinization as influenced by management and policy using two case studies. The first is representative of a developing country and explores management of salt-affected soils in the Indus Valley, Pakistan, while the second looks at a developed country, and illustrates how through monitoring we can deduce

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causes of shallow aquifer salinity in the Namoi Catchment of NSW, Australia. Finally, there is a section on integration and conclusions where we illustrate how management to mitigate salinization needs to be integrated with policy to diminish the threat to productivity that occurs with groundwater degradation.

15.1 Introduction

Globally the increased dependence on groundwater to maintain societies and their economies is mediated by threats to supplies of groundwater from a range of environmental and economic pressures, including depletion of supplies (Chap. 2), degradation of the water quality (Chaps. 2 and 14) and the energy issues associated with groundwater extraction and usage (Chap. 4). The criticalities and potential impacts of poorly-managed water resources are nowhere more divisive than where the balance between surface water and groundwater fluxes are upset and excessive amounts of salt are concentrated at the surface and in the shallow sub-surface. This can be caused both by excessive use of water, infiltrating to recharge shallow aquifers that fill to the surface where evapotranspiration concentrates salts in the near-surface, or through inadequate supply of water which does not flush salts beyond the root zone, hence also salinizing the sub-surface. This chapter addresses the degradation of water quality as it relates to salinization of resources and in particular the environmental degradation that occurs as a result of salinization processes. This degradation from salinization can be due to a combination of natural and anthropogenic processes, but these can be closely related. Specifically, this chapter addresses salinization as it affects agricultural productivity and does not consider naturally saline lands, though the consequence of anthropogenic mistreatment of landscapes containing salt stores can result in a similar situation that can prove extremely difficult to rectify and may require timeframes that prohibit economic recovery.

A report by FAO in 2000 estimated that globally the area of salt-affected soils including saline and sodic soils was 831 million ha (Martinez-Beltran and Manzur 2005). It extended over all the continents including Africa, Asia, Australasia and the Americas. This salinization results from the accumulation of water soluble salts in the upper layers of the stratigraphy. It has a major impact on agricultural productivity, environmental health and economic welfare. These salt stores in the stratigraphy can also cause increases in the salinity of groundwater, as salts can be mobilised through irrigation, deep drainage and recharge events. Thus the salinization of surface soils and groundwater supplies are intimately related.

This chapter will first describe the physical and chemical processes causing salinization of the root-zone and shallow aquifers, highlighting the dynamics of these processes and how they can be influenced by irrigation and drainage practices and thus illustrate the connectivity between soil and groundwater salinization. Conceptual diagrams will be used to depict fluxes of water and salt between the different compartments of the integrated system. Two case studies will then illustrate the extent of the problem of groundwater salinization as influenced by management and policy. The first is representative of a developing country and explores management of salt-affected soils in the Indus Valley, Pakistan, while the second looks at a developed country, and illustrates how through monitoring we can deduce causes of shallow aquifer salinity in the Namoi Catchment of NSW. Finally, this chapter will show how management to mitigate salinization needs to be integrated with policy to diminish the threat to productivity and groundwater degradation.

15.2 Major Types of Soil Salinity Based on Groundwater and Soil Processes

Three major types of salinity may be identified globally, determined by relative interactions between soil and groundwater processes (Rengasamy 2006), as shown in Fig. 15.1. Thus, these types include:

1. Groundwater associated salinity (GAS) where fluctuations in shallow ground-water levels lead to salt discharge into root zone layers
2. Non-groundwater associated salinity (NAS) caused by poor leaching due to restricting hydraulic properties of some soil layers (also referred to as transient salinity), and
3. Irrigation associated salinity (IAS) which is due to the input of salts in the irrigation water and their accumulation in the root zone due to inadequate drainage.

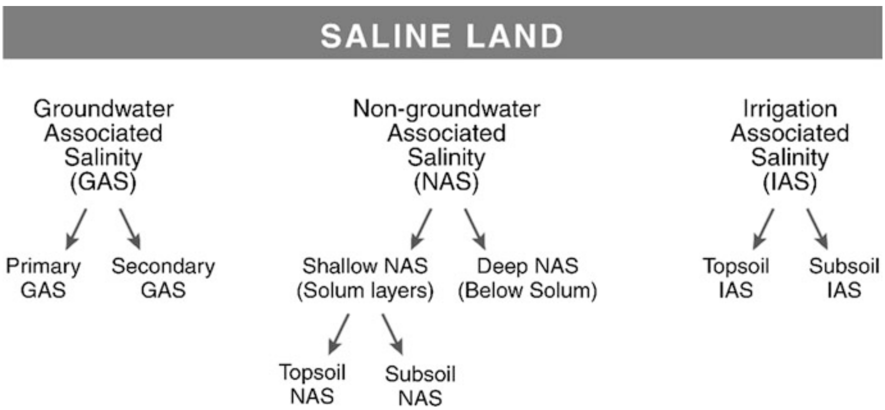

Fig. 15.1 Major types of salinity based on salinization processes

Irrigation associated salinity may also arise from excessive leaching of applied water that causes a rise in water tables and subsequent salt discharge and thus presents as groundwater associated salinity.

15.2.1 Groundwater Associated Salinity (GAS)

Salt accumulation in soil layers occurs when the water tables are shallow, particularly when they are <4 m from the surface and the salinity of groundwater becomes progressively higher due to evapotranspiration. Usually this situation occurs in foot slopes and valley floors of the landscape. Human activities, such as clearance of native deep-rooted perennial vegetation and subsequent agricultural practices, disturb the equilibrium levels of the water tables allowing increased recharge and salt movement from upper regolith layers that increases the salinity of the groundwater. Salts ultimately reach the surface via the discharge zone through capillary rise and high salinity levels in the soils can develop that are not conducive for agricultural production. Figure 15.2, for example, depicts an area of agricultural land in the Boorowa region of S.E. NSW badly affected by GAS.

15.2.2 NAS and Transient Salinity

Salt accumulation in root zone layers where water tables are deep, is termed transient salinity and generally salt concentrations fluctuate due to soil processes

Fig. 15.2 Agricultural land in SE NSW badly affected by GAS

and seasonal variability. Several sources of salts (as outlined in subsequent sections) contribute to the salts in the soil profile and these are concentrated due to evapotranspiration and the lack of sufficient leaching. The low hydraulic conductivity of many soil layers, which commonly occur in sodic subsoils, leads to poor drainage. Salt levels can be moderate to high and depend strongly on local soil and environmental conditions. Soil management and drainage options thus have to be specific for each site.

15.2.3 Irrigation Induced Salinity

The quality of irrigation water determines the amount and composition of salts which are stored in soil layers, while the relative hydraulic conductivity of the soil profile will determine the time taken for a specific area to salinize due to insufficient leaching. Irrigation management and drainage options are therefore also generally site specific.

15.3 Physical and Chemical Processes Causing Salinization of Root Zone Layers and Aquifers

While the processes of root zone and shallow aquifer salinization are inter-related, they have nevertheless traditionally been treated separately by agronomists and hydrogeologists, respectively. While attempts have been made to integrate our collective knowledge, the general disparity in ultimate drivers (agricultural productivity for agronomists and water supply for hydrogeologists) and scales of operation (<2 m depth at paddock scale for the former and >2 m depth and catchment scale for the latter) results in differing approaches; however these approaches ultimately converge where soil profiles and aquifers intersect. It is therefore instructive to approach salinization from these two directions to determine the intersection and synergies that exist.

15.3.1 Soil Processes and Salt Accumulation in the Root Zone

Accumulation of soluble salts above a certain level in the root zone of agricultural soils interferes with the crop production by either directly affecting the physiological functions of the plants and/or indirectly by disturbing soil physical and chemical conditions. The commonly used terms 'Primary' and 'Secondary' salinity are based on whether salt accumulates by natural phenomena or as a consequence of mismanagement of natural resources (viz. soil and water). There are several sources of salts causing soil salinization including natural weathering of soil minerals, salts added through precipitation (e.g. Blackburn and McLeod 1983) and salts associated with aeolian dust (e.g. Shiga et al. 2011). Other natural salinization processes include discharge of naturally saline groundwaters and saline groundwater intrusion. Agronomic practices such as fertiliser and pesticide application will also add salts to the soils, as will irrigation and dumping of waste materials. In addition to requiring a source of salts, climatic, hydrological and landscape features, combined with human activities and plant interactions, determine the specific location of salinization in the root zone and also the quantity and quality of salts accumulating.

15.3.1.1 Salinization of the Root-Zone

Figure 15.3 illustrates typical processes leading to salt accumulation in the root zone of a sodic soil, including the specific case of development of transient salinity (Fig. 15.3b). These salt stores in the root zone can have major effects on plants growth and soil processes (Fig. 15.3a) and can also affect groundwater supplies deeper in the landscape if recharge conditions change and a net downward flux of water and hence salt occurs.

15.3.1.2 Effects on Plant Growth

Irrespective of how the salts have accumulated in soil layers, the concentrations of soluble salts affect plant growth through the osmotic and ionic effects (Munns and Tester 2008). Thus, as salinity increases, the osmotic potential required to extract water into plant cells decreases and inhibits the water uptake by plants (water always moving from a higher to lower potential energy level). Increasing salinity

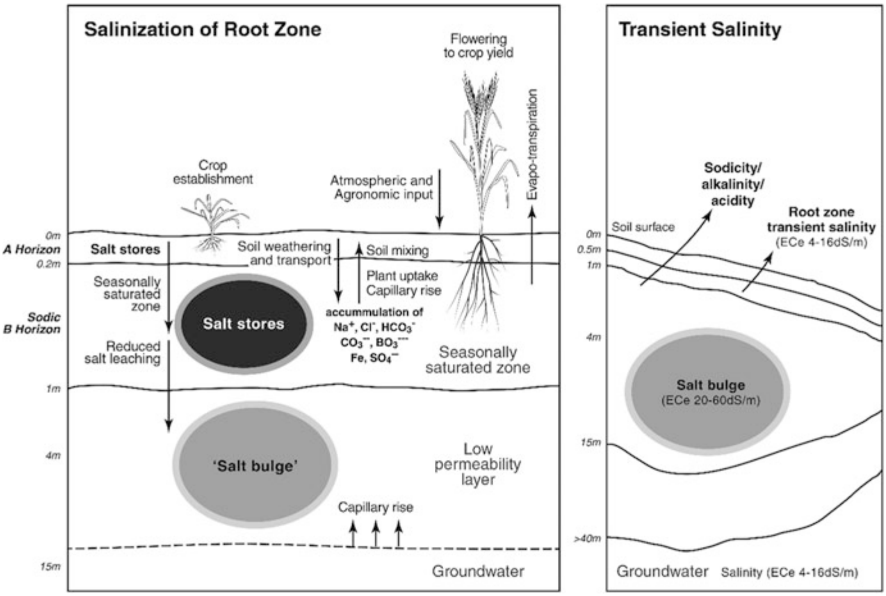

Fig. 15.3 (**a**) Salt accumulation in the root zone and effects on plant growth and soil processes. (**b**) Development of transient salinity (After Rengasamy 2006)

also leads to accumulation of ions in the plant over a period of time and leads to ion toxicity or ion imbalance.

15.3.1.3 Effects on Soil Processes

Salts can also affect many soil processes, such as soil water dynamics, soil structural stability, and solubility of plant nutrients under different hydrogen ion concentrations (pH) and electron conditions (Eh, or redox) of the soil water. Different categories of salt affected soils can therefore be distinguished based on the ionic composition of soil solution, each affecting soil properties and the mechanism of plant growth in different ways (Table 15.1).

15.3.1.4 Effect of Cations and Anions on Soil Structure

It has been well established that when the amount of soil adsorbed sodium ions, commonly measured as exchangeable sodium percentage (ESP) in soils (or estimated through the sodium adsorption ratio (SAR) of the soil solution), increases above a critical value (ESP > 6 in Australia and ESP > 15 in USA and other countries), and the EC of the soil solution decreases below a critical threshold value, the soil structure deteriorates severely due to dispersion of the clay micro-aggregates or quasi-crystals (Greene et al. 1973; Quirk and Schofield 1955; Rengasamy et al. 1984). With clay dispersion and concomitant blockage of the soil pores, permeability is reduced; this effect is particularly pronounced in smectite dominant clay soils (Turner et al. 2008). As a consequence water and air movement and water storage are highly restricted thereby affecting root environments and, consequently, plant growth (Rengasamy 2013). SAR is readily determined from the major cation composition of soil water and hence used as an indicator of sodium effects on soils.

$$\mathrm{SAR} = \mathrm{Na}^{+} / \left(\mathrm{Ca}^{2+} + \mathrm{Mg}^{2+}\right)^{0.5}$$

where the concentrations of Na^{+}, Ca^{2+} and Mg^{2+} in soil solutions are expressed as $\mathrm{mmol.L}^{-1}$.

SAR, however, has a number of limitations which distil its effectiveness in predicting soil effects. Potassium (K^{+}) ions, for example, are not considered in this SAR model, even though adsorbed potassium has been found to affect soil structure (Rengasamy and Sumner 1998). This is partly because K^{+} has traditionally been hard to quantitatively measure and fortunately has been found to occur at relatively low concentrations (<5 %) in most waters. Recently, Marchuk and Rengasamy (2011) derived ionicity indices of clay-cation bonding and showed that the dispersive effects (which causes soil structural problems) of Na^{+} and K^{+}, and the flocculating effects (which promote soil structural integrity) are highly related to ionicity, or covalency, of the clay-cation bondings.

Further, the effects of Ca^{2+} and Mg^{2+} in reducing monovalent adsorption are considered to be equal in the SAR equation. However, Rengasamy and Sumner

Table 15.1 Categories of salt-affected soils based on EC_e (dS/m), SAR_e and $pH_{1:5}$ of soil solutions and possible mechanisms of impact on plants. Toxicity, deficiency or ion-imbalance due to various ions will depend on the ionic composition of soil solution

No.	Category of saline soil	Criteria	Possible mechanisms of impact on plants
1	Acidic-saline soil	$EC_e > 4$; $SAR_e < 6$; pH < 6	Osmotic effect; microelement (Fe, Al, Mn etc.) toxicity; SO_4^{2-} toxicity in very low pH
2	Neutral saline soil	$EC_e > 4$; $SAR_e < 6$; pH 6–8	Osmotic effect; toxicity of dominant anion or cation other than Na^+
3	Alkaline-saline soil	$EC_e > 4$; $SAR_e < 6$; pH 8–9	Osmotic effect; HCO_3^- and CO_3^{2-} toxicity;
4	Highly alkaline-saline soil	$EC_e > 4$; $SAR_e < 6$; pH > 9	Osmotic effect; HCO_3^- and CO_3^{2-} toxicity; microelement (Fe, Al, Mn etc.) toxicity
5	Acidic-saline-sodic soil	$EC_e > 4$; $SAR_e > 6$; pH < 6	Osmotic effect; Na^+ and microelement (Fe, Al, Mn etc.) toxicity
6	Neutral saline-sodic soil	$EC_e > 4$; $SAR_e > 6$; pH 6–8	Osmotic effect; Na^+ toxicity; toxicity of dominant anion (Cl^- or SO_4^{2-})
7	Alkaline-saline-sodic soil	$EC_e > 4$; $SAR_e > 6$; pH 8–9	Osmotic effect; Na^+ toxicity; $HCO3^-$ and CO_3^{2-} toxicity
8	Highly alkaline-saline-sodic soil	$EC_e > 4$; $SAR_e > 6$; pH > 9	Osmotic effect; Na^+ toxicity; $HCO3^-$ and CO_3^{2-} toxicity; microelement (Fe, Al, Mn etc.) toxicity
9	Acidic-sodic soil	$EC_e < 4$; $SAR_e > 6$; pH < 6	Indirect effect due to soil structural problems; seasonal waterlogging can induce microelement (Fe, Al, Mn etc.) toxicity
10	Neutral sodic soil	$EC_e < 4$; $SAR_e > 6$; pH 6–8	Indirect effect due to soil structural problems; seasonal waterlogging; Na^+ toxicity at high SAR_e
11	Alkaline-sodic soil	$EC_e < 4$; $SAR_e > 6$; pH 8–9	Indirect effect due to soil structural problems; seasonal waterlogging; Na^+ toxicity at high SAR_e; HCO_3^- and CO_3^{2-} toxicity
12	Highly alkaline-sodic soil	$EC_e < 4$; $SAR_e > 6$; pH > 9	Indirect effect due to soil structural problems; seasonal waterlogging; Na^+ toxicity at high SAR_e; HCO_3^- and CO_3^{2-} toxicity; microelement (Fe, Al, Mn etc.) toxicity

(1998) showed the flocculation efficiency of Mg is actually about 0.6 times that of Ca.

Based on these recent discoveries, Rengasamy and Marchuk (2011) proposed an alternative "cation ratio of soil structural stability" (CROSS) to replace SAR. CROSS incorporates the differential effects of Na^+ and K^+ in dispersing soil clays, and also the differential effects of Ca^{2+} and Mg^{2+} in flocculating soil clays. CROSS is defined as:

$$\mathrm{CROSS} = \left(\mathrm{Na}^{+} + 0.56\ \mathrm{K}^{+}\right) / \left(\mathrm{Ca}^{2+} + 0.6\ \mathrm{Mg}^{2+}\right)^{0.5}$$

where the cation concentrations are expressed in $\mathrm{mmol.L}^{-1}$.

CROSS estimated in soil solutions can be used as a better indicator of potential soil structural effects when compared to SAR (Marchuk and Rengasamy 2012; Marchuk et al. 2013). The actual effects on soil structure depend on the total electrolyte concentration, generally measured as electrical conductivity (EC). When the ionic strength, or EC, of the soil solution is below a critical threshold level (see Table 15.1), the cationic effects on soil structure are predominant.

Anions, even in low ionic strength solutions, critically contribute to the pH of the soil solution. This alters the net charge on soils and thereby affects the influence of cations. Thus, sulphates and chlorides tend to contribute to acidic pH levels while bicarbonate and carbonate ions generally promote alkaline conditions. Notably, the dispersive effects of Na^{+} and K^{+} are enhanced when the soil pH is above 8.5 (Marchuk et al. 2013).

15.3.1.5 Management of Salt-Affected Soils

Sustainable agriculture in salt-affected soils, whether in dryland or irrigated regions, will depend on maintaining low levels of salinity as well as maintaining the proper balance of cations and anions. Accurate, spatially explicit data on soil water chemistry is vital and consideration of the potential effects of differing sources and expressions of salinity is required to make informed assessment of appropriate remediation strategies. While application of fresher waters may serve to dilute salt-affected lands, if the inherent conditions are highly sodic, or have high acidity, or alkalinity, then reduction in salinity alone my result in further degradation of soil condition (Turner et al. 2008) (Table 15.1).

15.3.2 Processes Leading to Salinization of Aquifers

15.3.2.1 Aquifer Salinization in Inland Areas

Foster and Chilton (2003) have summarised decades of investigations into salinization of aquifers and consolidated several processes leading to salinization of aquifers in inland areas (Fig. 15.4). These processes include: (i) rising groundwater tables due to inefficient surface irrigation and inadequate drainage, (ii) natural salinity mobilized from the landscape due to land clearing of native vegetation, and (iii) disturbance of natural groundwater salinity stratification by well construction and groundwater extraction. Some of these processes of groundwater salinization are discussed in more detail in the case studies below of (i) the management of salt-affected soils in the Indus Valley, Pakistan, and (ii) the monitoring and investigation of the causes of shallow aquifer salinity in the Namoi Catchment of NSW.

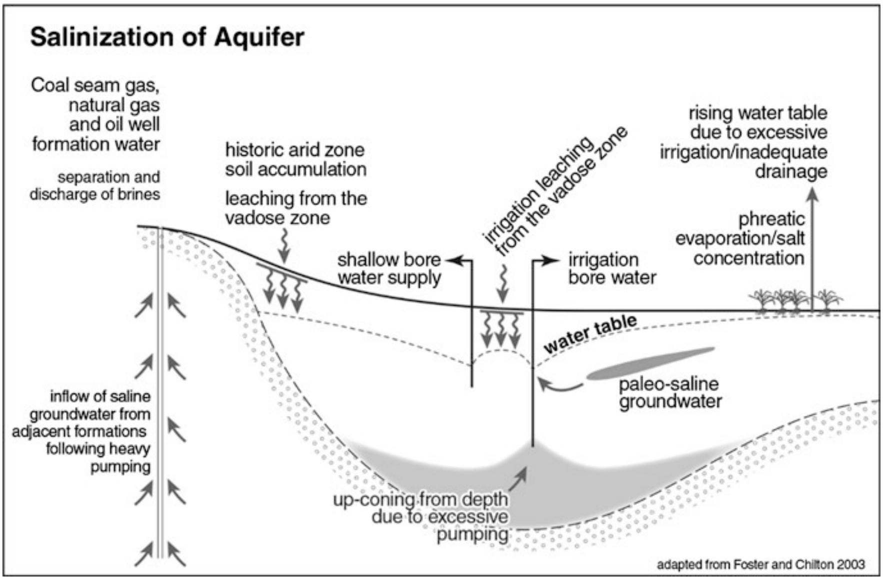

Fig. 15.4 Processes leading to salinization of aquifers in inland areas (After Foster and Chilton 2003)

15.3.2.2 Aquifer Salinization in Coastal Areas Due to Intrusion of Salt Water

Coastal floodplains around the globe constitute prime agricultural areas that generally rely on conjunctive use of surface water and groundwater, and this is largely controlled by the strong seasonality expressed in surface water supplies. As agricultural productivity has increased, traditional seasonal, opportunistic irrigation has been replaced by year-round development with seasonal surface waters augmented by groundwater during low-flow periods. Excessive pumping during groundwater extraction in coastal areas, however, can lead to salinization due to induced sea water intrusion. The hydraulic head of inland groundwaters is reduced by excessive pumping allowing seawater to encroach further inland, and thus salinizing the landscape (Fig. 15.5).

Once seawater intrudes and causes coastal salinization, it is almost impossible to remediate. Salinization of fresh groundwater in coastal aquifers is a global issue that is exacerbated by excessive groundwater extraction as well as by sea level rise (Werner et al. 2013). Under natural hydraulic equilibrium, a sloping interface between fresh and saline pore waters within an aquifer is located beneath the coastal plain (Fig. 15.5). Groundwater extraction at rates exceeding up-stream recharge by freshwater allows the interface to progress inland and locally may cause increased upwards and landward flow of saline seawater. The natural groundwater equilibrium is further susceptible to changes in recharge and discharge caused by climate change. Khan et al. (2006) have discussed how increasing rates

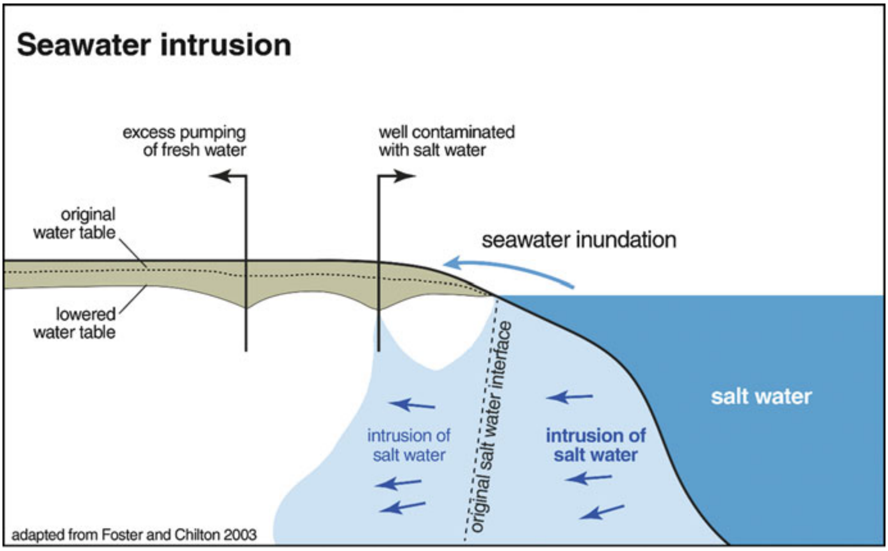

Fig. 15.5 Process leading to salinization of fresh groundwater in coastal aquifers due to the intrusion of salt water from the sea

of sea level rise due to global warming will increase the potential for intrusion of salt water in coastal areas of Bangladesh.

Critically, fresh water that is contaminated by only 5 % of seawater renders it unsuitable for many beneficial uses without treatment. In low lying areas, salinization of fresh coastal aquifers commonly also occurs by inundation (Fig. 15.5), where seawater floods across the surface during storm surges or tsunami or as the land surface subsides. Seawater inundation infiltrates through soil to underlying aquifers as unstable lobes of saline water. Wong et al. (2015) also described how storm surges and sea-level rise resulted in the short-term inundation of low level coastal floodplain sediments. The inundation by either brackish water or seawater results in a decline in surface water quality due to increase liberation of acidity and trace metals.

15.3.2.3 Aquifer Salinization Related to Extractive Resource Industries

Resource extraction industries may also affect groundwater salinity as they physically interfere with existing aquifers and rock formations that are saturated with groundwater. Commonly, groundwaters associated with mineral resources (e.g. coal, oil and gas) exhibit high salinity and appropriate management is required to mitigate any contamination of fresher water supplies, soils and the environment.

Although the volume of water used by mining is typically small relative to other users, such as agriculture and the environment, cumulative volumes of groundwater that may require disposal can be large and, if challenges in managing salinity can be adequately addressed, this produced water presents significant opportunities as an alternative water supply for communities, industry, and the environment, albeit potentially short-lived (Timms and Bourke 2014).

As an example, management of salt at coal mining operations in the Hunter Valley involves multiple strategies to protect fresh surface waters. Farmers, concerned that increasing river salinity was impacting the irrigation of food and fodder crops, worked together with the mining and power generation industries to develop a world leading salt trading scheme. The Hunter River salt trading scheme (HRSTS) involves 13 mines and 3 power stations that trade salt credits permitting controlled release of saline water to the river during high flow events (Selman et al. 2009; Vink et al. 2013). Discharge volumes, salt concentrations and dilution factors are continuously monitored to achieve salinity targets at key points along the river. Allowable maximum salinity in the river during discharge events is set as 900 μS/cm at the most downstream monitoring point of the scheme. Thus, mine-site contribution to the salt load in the Hunter River, via controlled discharges, was contained to 3 % of total river load under the HRSTS between 1995 and 2001 (David et al. 2003). Ultimately, however, if salt containment and dilution strategies are unable to achieve water quality objectives, active treatments such as desalination are required at some mine sites.

An additional salinity concern relates to final voids left following mine closure. These voids act as a point of groundwater recharge, or a hydraulic sink for local groundwater flow depending on local conditions. Evaporative concentration of saline groundwater in open voids often occurs, with the salinity of the void water increasing until salt solubilities are exceeded and precipitation occurs. This is a rare event however, as inputs of fresh runoff will delay the salinization. Geochemical modelling estimates of water quality in an open mine in the Hunter Valley, for example, indicates that it would take over 400 years for water salinity to exceed 4,000 mg/L (Hancock et al. 2005). A recent Environmental Impact assessment for a proposed open cut coal mine in the semi-arid Gunnedah Basin estimates 30,000 mg/L after ~420 years of evaporative concentration of inflowing groundwater at a salinity of 5,000 mg/L (Shenhua Watermark EIS 2013).

The recently developed water accounting framework for the minerals industry in Australia (MCA 2012), provides a common method to compare water balance and quality on a site-by-site basis and identifies three categories of water with respect to salinity to guide water use options, which may include dust suppression, industrial use or being suitable for potable supplies (Table 15.2).

Coal bed methane (CBM) operations, known as coal seam gas (CSG) in Australia, can generate large volumes of groundwater that may be quite saline and requires careful management to ensure that the risk of salinization to other groundwater and surface water supplies is negligible (Williams et al. 2012). Groundwater salinity could be impacted by CBM operations via a number of mechanisms including: mobilisation of saline groundwater through enhanced hydraulic gradients towards gas wells; leakage of saline groundwater through poorly sealed exploration bores and aging water supply bores; and surface spills from pipelines and storages for produced water and associated production fluids. Produced water volumes are typically highest during the early stages of CBM well

Table 15.2 Water types and salinity classes

Water type	EC (dS/m)	TDS (mg/L)	Mine Water[a] TDS (mg/L)
Pure rainwater	<0.015	<10	
Freshwater	0.015–0.8	100–1,000	<1,000 (Category 1)
Slightly brackish water	1.6–4.8	1,000–3,000	1,000–5,000 (Category 2)
Brackish water	4.8–16	3,000–10,000	>5,000 (Category 3)
Saline water	16	>10,000	
Seawater	51.5	35,000	
Hyper saline	>51.5	>35,000	

[a]MCA 2012

development as the system is depressurised to allow gas to desorb from the coal matrix. Risks to groundwater quality can typically be managed to reduce the risk of impacts to very low or negligible issues, although some concerns remain to be addressed. Disposal via evaporation and seepage ponds are no longer permitted in eastern Australian states and thus removes one of the highest risks to aquifer salinization. Temporary storage ponds, however, are still required and must be designed to minimise the risk of spills and seepage losses.

The salt content in produced CBM water varies widely, from nearly freshwater (10–500 mg/L) to salt levels up to ten times higher than seawater (300,000 mg/L). Lower concentrations tend to be associated with shallow coal seams exposed to recent fresh surface water recharge (Khan and Kordek 2014). In the Sydney Basin, existing CBM operations currently produce <4.5 ML/year of water with a salinity between 7 and 15 dS/m (i.e. TDS of 4,700–10,000 mg/L). The produced water is reused in drilling operations and the excess treated at a licensed water treatment facility. By contrast, CSG operations in the Surat Basin in South-east Queensland are projected to produce 20,000 ML/year of water for 50 years with a TDS of 14,500–31,000 mg/L (OGIA 2012).

Produced water that is in excess of operational requirements will generally be treated by reverse osmosis desalination with discharge of suitably treated water for beneficial use. Beneficial use of mixed or treated water, for example to augment irrigation and environmental flows to rivers, is currently encouraged by regulatory agencies. Table 15.3 indicates water salinity limits for selected beneficial uses in Australia for irrigation of crops and for stock water. Water that is not suitable for drinking water (>1.2 dS/m) for example, is fit for the purpose of irrigating crops that are tolerant to brackish water.

Brine produced by desalination would be concentrated and recrystallized, typically for disposal to landfill. An alternative method of disposal of saline produced water by deep well injection could provide a local and permanent disposal solution for produced water or brine concentrates (National Research Council 2010). Commonly practiced in some parts of the US, deep well injection targets naturally saline formations that are hydraulically disconnected from fresh water aquifers (Yeboah and Burns 2011).

Table 15.3 Salinity guidelines for key beneficial uses

Irrigation	EC (μS/cm)	Comments	Source
	8,000	Unsuitable for barley irrigation	ANZEEC (2000)
	7,700	Unsuitable for cotton irrigation	
	5,500	Unsuitable for sunflower irrigation	
	6,000	Unsuitable for wheat irrigation	
	1,500	If used on early season cotton, the final yields could be diminished	
Livestock	14,920[a]	Loss of production and a decline in beef cattle condition and health	
	10,450[a]	Loss of production and a decline in dairy cattle and horses condition and health	
	11,940[a]	Loss of production and a decline in pigs condition and health	
	5,970[a]	Loss of production and a decline in poultry condition and health	
	19,400[a]	Loss of production and a decline in sheep condition and health	
Drinking water[b]	<120[c]	Excellent drinking water quality	ADWG (2008)
	120–750[c]	Good drinking water quality	
	750–1,200[c]	Fair drinking water quality	
	1,200–1,490[c]	Poor drinking water quality	
	>1,490[c]	Unacceptable drinking water	

[a]Note that if the TDS concentration is above 2400 mg/L, the water should be analysed to determine the concentrations of specific ions to avoid possible toxication (ANZEEC 2000)
[b]Bruvold and Daniels (1990) in Australian Drinking Water Guidelines (2008)
[c]TDS values converted to EC using equation: EC (μS/cm) × 0.67 = TDS (mg/L) (ANZEEC 2000)

15.3.2.4 Hydrogeochemical Changes During Salinization

Examination of the changing proportions of dissolved ions in groundwater can reveal more information about the processes that lead to increased salinity or EC. Mixing and geochemical reactions that are common along a flow path change specific ion concentrations, or hydrochemical facies. Water recharged from meteoric sources is typically represented by fresh, bicarbonate-type water with mixed cation composition. Geochemical interactions between water and sediment, or rock, along a flow path typically results in evolution towards either a sodium bicarbonate water or sodium chloride water. Hypothetically, a series of hydrochemical facies occurs along a flow path, depending on the nature of sediment and rock encountered, that can indicate the maturity of a groundwater system.

A Piper diagram, as shown in Fig. 15.6, is a common method by which hydrochemical facies are represented. Relative concentrations of major cations are plotted in the lower left hand ternary plot, and relative concentrations of major anions are plotted in the lower right hand ternary plot. Both these sets of points are then projected to the central diamond, to intersect at a point that is indicative of hydrochemical facies. For example, Fig. 15.6 depicts facies change

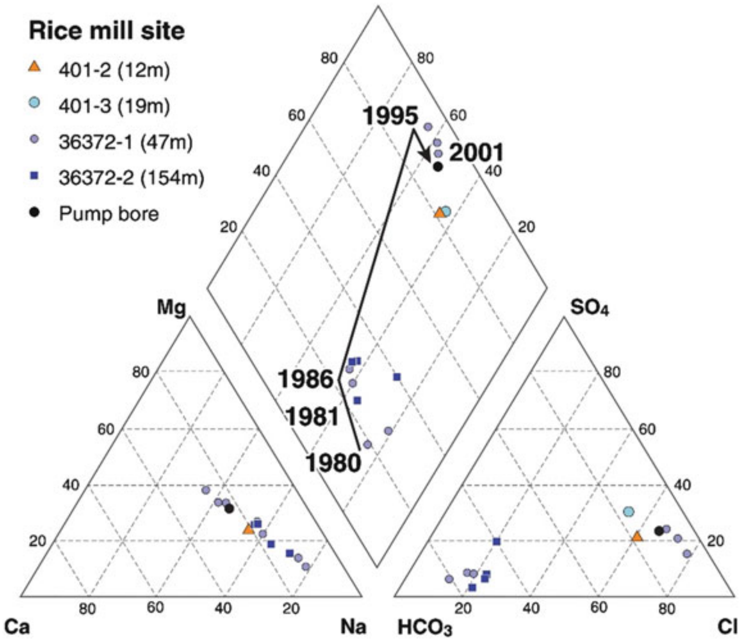

Fig. 15.6 Piper diagrams showing changes in major ion composition of groundwater at the Rice Mill site between 1980 and 2001 (After Timms 2001)

during salinization over two to three decades at a site (Rice Mill) in the Murrumbidgee Irrigation Area (MIA) of Australia's Murray-Darling Basin. At this site, groundwater in alluvial sediments at a depth of approximately 50 m increased in EC from <0.5 dS/m to between 1.3 and 4.1 dS/m. Concomitant with this change in EC, bicarbonate type water changed towards a chloride type water. The major ion composition of the deep groundwater also trended towards a similar composition, though less pronounced, and this was attributed to a downwards hydraulic gradient caused by groundwater extraction from the underlying aquifers, causing the leakage of groundwater from salt laden shallow sediments (Timms and Acworth 2002).

The primary geochemical reactions include dissolution of salts; ion exchange with clay minerals and surface sorption and desorption reactions. These geochemical reactions proceed towards chemical equilibrium and can therefore be assessed using models based on thermodynamic chemical databases. A simple approach utilises a water mixing diagram that can be developed prior to geochemical modelling based on conservative ion (e.g. chloride) concentrations of end members such as saline and fresh groundwater. Figure 15.7, for example, illustrates that at high mixing ratios, based on conservative ion mixing, HCO_3 and SO_4 concentrations are non-linear indicating processes other than mixing controlling concentration. A relative increase of Na in fresh water samples is evident,

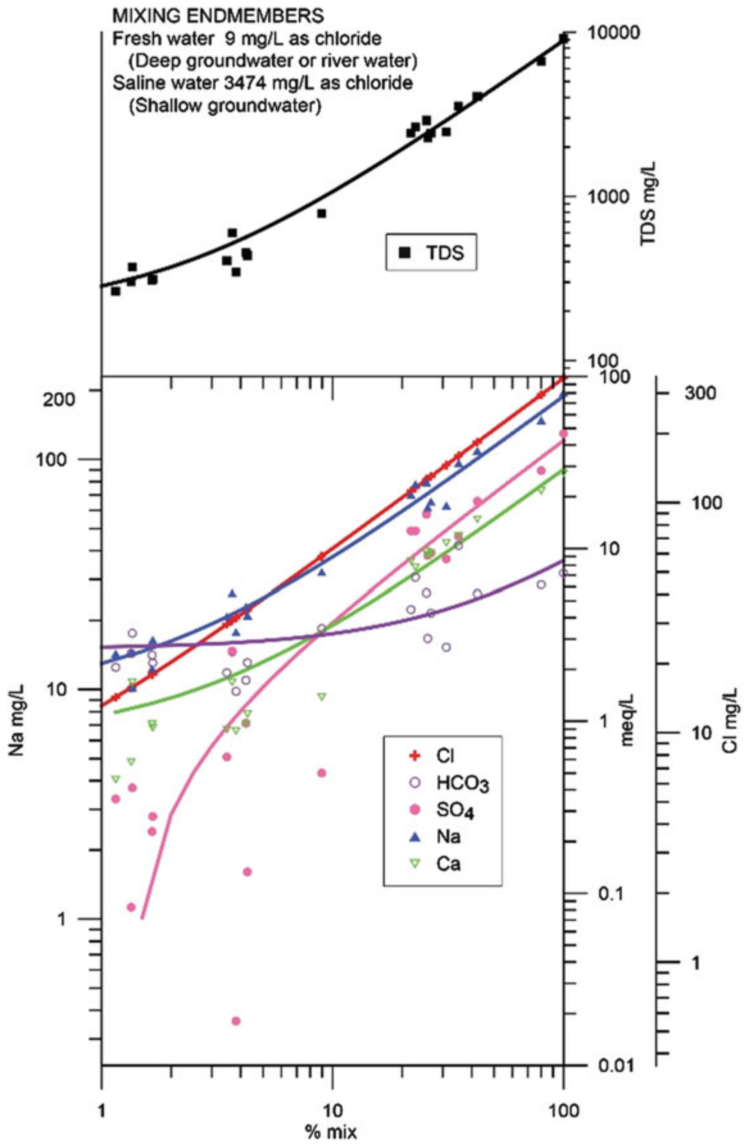

Fig. 15.7 Mixing diagram for groundwaters (N = 44) arranged along the X axis in terms of % mix of saline groundwater of 3,474 mg/L chloride in fresh water of 9 mg/L chloride (After Timms 2001)

indicating ion-exchange is an important geochemical process that occurs during mixing. Incorporation of geochemical mixing models to help understand salinization processes can aid in determination of appropriate mitigation strategies and management controls.

15.4 Salinization Case Studies

This section illustrates the scale of the problem of groundwater salinization as influenced by management and policy through two case studies in a developing and a developed country, i.e. Pakistan and Australia, respectively. The case study from Pakistan focuses mainly on soil salinization, whilst the case study from the Namoi catchment particularly assesses aquifer salinization.

15.4.1 Case Study 1: Rehabilitation of Salt Affected Lands in the Wheat-Cotton Zone of Pakistan; a Physical and Economic Approach to Water Logging and Irrigation Salinity

15.4.1.1 Overview

In Pakistan approximately 80 % of the agricultural production comes from irrigated agriculture. However, waterlogging and irrigation salinity are major land degradation problems that result in severe economic and social consequences. Out of the total 16.3 million ha of irrigated land in Pakistan about 6.2 million ha (38 %) are waterlogged, and 2.3 million ha (14 %) are saline. The irrigation salinity results from the intensive use of surface irrigation, where the salt accumulates in the soil surface layers due to a major imbalance in the amount of salt entering and leaving the soils. As a consequence yields and production of crops are adversely affected, resulting in severe economic losses. In 2001 these losses were estimated to be 350 million US$ annually. Thus salinity management can offer opportunities to alleviate poverty and improve rural livelihoods. Economically, a viable choice of the salinity management is needed to guide decisions for future salinity investments.

Eastern Sadiqia South region in the wheat-cotton zone of Punjab province has been used as a case study. The aim of this study was to (i) conduct a cost benefit analysis of implementing three management strategies, i.e. no intervention, an engineering approach, and an agronomic approach, on four land types, and (ii) compare the Net Present Values (NPV) of the three strategies on the four land types over a period of 25 years, using a discount rate of 6 %. The results of the study will enable us to define criteria and set out rules for investment in the rehabilitation of salt-affected lands.

15.4.1.2 Background

The Fordwah canal command area of Eastern Sadiqia South (FESS) region is located in the wheat-cotton zone of Punjab province, and forms part of the Indus Basin irrigation system. The area is situated on the left bank of the River Sutlej (Fig. 15.8).

The climate of the regions is arid (PARC 2002), with annual evaporation of 2,000 mm, far in excess of the annual rainfall of 240 mm (Sarwari et al. 2000). The soils of the area are comprised mainly of a wide range of coarse to fine textured

Fig. 15.8 Location of Eastern Sadiqia region, the study area (Source: Pakistan Agriculture Research Council 2002)

alluvial deposits, dominated by medium textured silty loams, with a low to medium water holding capacity (Sarwari et al. 2000). In the majority of cases, farmers use flood irrigation (Kijne 1996), sourced from canal water with an average EC value of 0.3 dS/m (IWMI 2007). Groundwater in most of the region, however, is saline with EC > 4 dS/m (Aslam and Prathapar 2006), making groundwater mostly unsuitable for irrigation purposes. Deep percolation from the irrigation system of canals, combined with general lack of good drainage, has resulted in water logging and a major imbalance in the amount of salt entering and leaving the soils. The water logging and salt accumulation have caused various degrees of degradation (Fig. 15.9) (Kahlown and Azam 2002).

Four different land categories were identified (Fig. 15.9) based on the severity of water logging (depth of water table) and soil salinity (ECe) (Table 15.4). They ranged from Land Class D1 which was extremely degraded land through to Land Class 4 which was normal land. Three different salinity management strategies were investigated, i.e. 1. no intervention and use of existing land-use cropping activities, 2. an engineering approach where excessive water is drained from the irrigated land, and 3. an agronomic approach whereby crop diversification, e.g. kallar grass and sesbania, and planting of deep rooted vegetation, e.g. River Red Gum, were used.

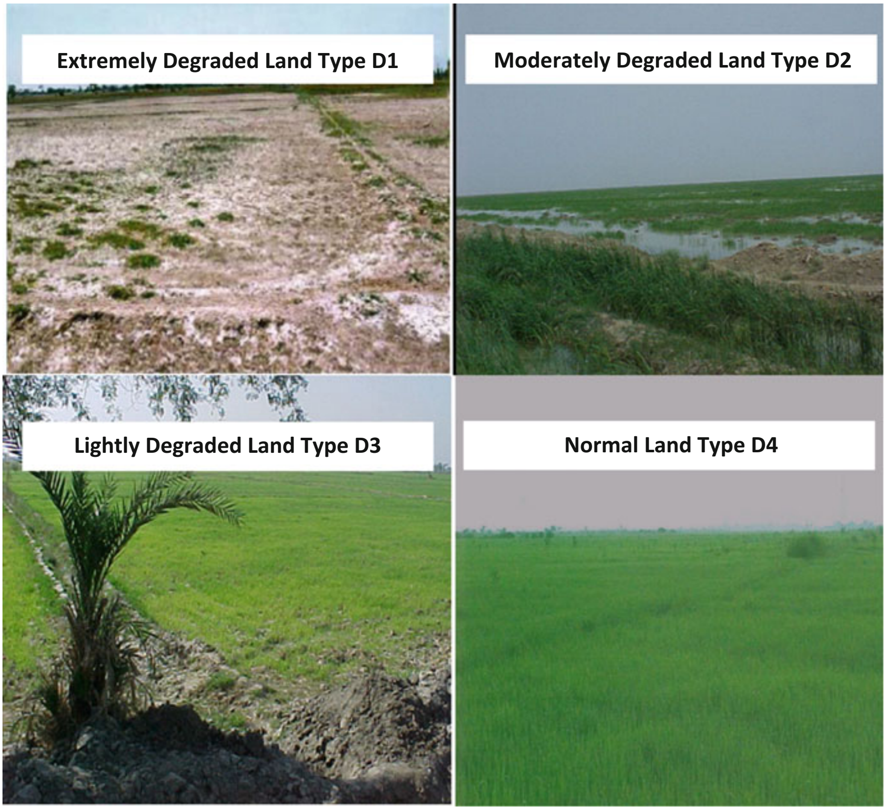

Fig. 15.9 Land types of the Eastern Sadiqia region (Compiled from IWMI 2007)

Table 15.4 Criteria for land categories in the Eastern Sadiqia (FESS) region (Adapted from PARC 2002)

Land category	Description
D1 Extremely degraded land	Watertable < 1 m and ECe >12 dS/m
D2 Moderately degraded land	Watertable 1–2 m and ECe between 8 and 12 dS/m
D3 Lightly degraded land	Watertable 2–3 m and ECe between 4 and 8 dS/m
D4 Normal land	Watertable >3 m and ECe between 0 and 4 dS/m

15.4.1.3 Physical Analysis of Effects of Salinity on Soils and Crops

A salt balance model proposed by Hillel (2000) (Fig. 15.10) was used to calculate the rate of salt accumulation under the four land categories. The model identifies five sources of salt inputs: the salt already present in the mineral soils; the salt entering the root zone from irrigation water; salt addition through land management practices such as fertilizers; salt addition through rainwater; and the salt entering

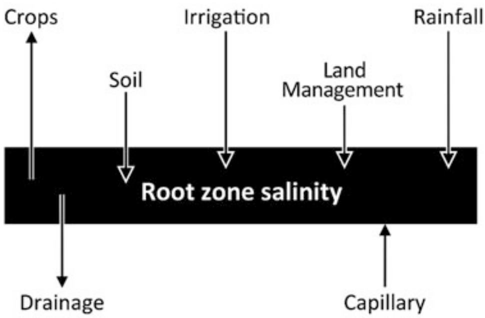

Fig. 15.10 Salt balance calculation model proposed by Hillel (2000)

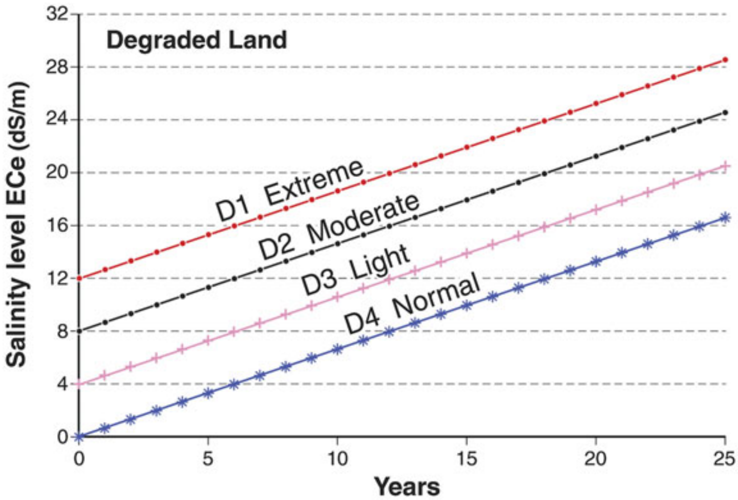

Fig. 15.11 Salinity projections for the next 25 years in the study area

the root zone from groundwater through capillary rise (Fig. 15.10). The factors of salt removal include removal by the crops and by natural drainage.

Figure 15.11 indicates the increasing trend of land salinity for the next 25 years in the four major categories of land (D1, D2, D3, and D4) present in the study area under irrigated agriculture and with no management strategy in place. The annual rate of salt accumulation was calculated to be 0.420 kg/m^2 or 0.66 dS/m (Arshad 2007). Wheat and cotton crop yields start to decrease with the increase in the salinity levels for each year. Figure 15.12 shows the trend of wheat and cotton crop yields under the influence of salinity. There is no effect of salinity on both crops up to an ECe of 4 dS/m; however, after an ECe of 4 dS/m, yield of both crops show a rapid decrease with increase in ECe, with wheat being the most affected.

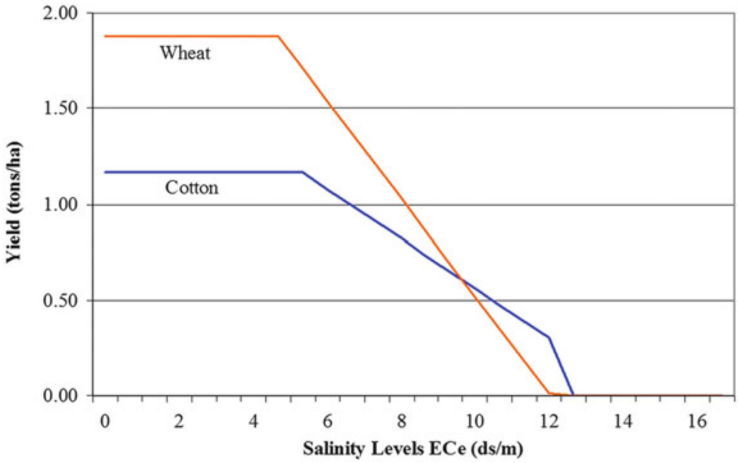

Fig. 15.12 The effect of increasing salinity levels on wheat and cotton yields in the study area

Table 15.5 Net present values (NPV) for three salinity management strategies (US $/ha)

Land type	No management strategy	Engineering strategy	Agronomic strategy
D1 Extremely degraded land	−408	673	1,078
D2 Moderately degraded land	57	1,042	477
D3 Lightly degraded land	1,544	1,074	477
D4 Normal land	3,500	49	95

15.4.1.4 Results of the Cost Benefit Analysis

By considering the four land types (D1, D2, D3, and D4) this desktop study compares the Net Present Values (NPV) of the two diverse management strategies, of engineering and agronomic options, with the no intervention strategy. Table 15.5 highlights the rehabilitation strategies that maximise Net Present Values for the four land types over a period of 25 years. A discount rate of 6 % was applied to all three strategies analysed.

The results of the cost benefit analyses in the study area indicate several important outcomes:

Firstly, for the extremely degraded land type D1, the agronomic strategy will result in the highest farm benefits in the next 25 years. Managing extremely degraded land type D1 with the engineering strategy ranked 2nd, and with the no management strategy ranked 3rd. Secondly, for the moderately degraded land type D2, the engineering strategy provides the best economic solution, when compared with the agronomic strategy which ranked 2nd, and the no management strategy which ranked 3rd. Thirdly, for the lightly degraded land type D3, no management is the best economic strategy. After the no management strategy, the engineering strategy ranked 2nd and the agronomic strategy ranked 3rd. Fourthly, for the normal land type D4, the no management strategy provided the highest benefits followed by the agronomic strategy which ranked 2nd and the engineering strategy which ranked 3rd.

One limitation of the previous analysis, where the criteria for the rehabilitation of salt affected lands are purely economic, is that it will encourage farmers to maximise profits at the cost of soil and land degradation. For example results in Table 15.2 suggest that the No Management Strategy for Land types D3 (NPV, 1,544 $/ha) and D4 (NPV 3,500 $/ha) is the best economic option. This implies that both the productive Normal and Lightly Degraded Lands under irrigation should be managed according to normal practice without any additional salinity management. However, following only this economic criteria, the two land types D3 and D4 will become extremely salinized by the end of years 12 and 18 respectively, with salinity levels EC_e exceeding 12 dS/m (Table 15.4 and Fig. 15.11). Thus a long term farm plan considering a ≥ 50 year scenario may encourage farmers to initially degrade the productive lands (D3 and D4) in the first 25 years and concurrently restore the degraded lands (D2 and D1) via an engineering and agronomic strategy respectively. However, in the second 25 year period, theoretically the initially productive lands (D3 and D4) which have now become degraded due to salinization, should undergo active rehabilitation via an engineering or agronomic strategy. Similarly land types D1 and D2 which were initially extremely and moderately degraded respectively and became productive after 25 years, should in the next 25 years be managed without any additional inputs.

In summary, this study serves as a decision support tool for funding agencies to undertake future investments in salinity management. The study informs two guiding principles for the salinity management policy in Pakistan: first, highly degraded lands such as D1 and D2 should be given priority for salinity rehabilitation (via an engineering and agronomic strategy respectively) over the relatively productive lands such as D3 and D4; and second a longer period of analysis of at least 50 years should be adopted for determining economic viability of salinity investment strategies. However even though a ≥ 50 year scenario may be adopted such an approach is still limited because it only considers economic criteria and doesn't take into account broader environmental factors such as (i) climate change, and (ii) the problem of how to dispose of the salt in the long term and how to prevent this salt contaminating other areas. Such factors may adversely impact on the long-term sustainability of irrigated agriculture in the region.

15.4.2 Case Study 2: Groundwater Salinity Changes in the Namoi Catchment

15.4.2.1 Overview

The salinity of alluvial aquifers in the Namoi catchment, located in the north-west of Australia's Murray-Darling Basin, has changed over the last several decades. Since the mid-1980s, groundwater from some bores has become more saline, while in others groundwater has become fresher (Badenhop and Timms 2012). Aquifer salinization processes in the Namoi catchment area could be attributed to several processes that mobilize salt from shallow soils and aquifers. These are investigated below.

15.4.2.2 Background

The Namoi catchment covers an area of approximately 42,000 km^2 and is located in northeast New South Wales in a semi-arid to arid setting with summer-dominant rainfall. In the Namoi catchment, ~112,000 ha of land is irrigated to grow cotton, wheat and other crops, using a combination of surface-water and groundwater supplies depending on water availability (CSIRO 2007). Groundwater resources occur primarily in semi-confined, alluvial aquifers up to ~100 m depth and exhibit complex hydraulic connectivity pathways (Kelly et al. 2014). Relatively low, fresh to saline, groundwater yields are achieved from multiple watertable alluvial aquifers with additional supplies from porous and fractured rock aquifers accessed on ridges above the black soil plains, and from beneath the alluvial aquifer system.

Low salinity groundwater is required for the environment, stock water and for irrigation in the Namoi Catchment, to support an industry worth at least $380 million each year. In addition, potable drinking water supplies are sourced almost exclusively from groundwater in this semi-arid region. Continued extraction has resulted in a clear trend of falling groundwater levels, with up to 14 m drawdown in some areas since the beginning of the groundwater withdrawals in the 1960s, with the growth of the irrigation industry and development of groundwater management approaches in the area and response to historic and recent flooding (Kelly et al. 2013).

15.4.2.3 Groundwater Salinity Changes over Time

In the late 2000s, the Namoi Catchment Management Authority (CMA) commissioned a study to evaluate groundwater salinity changes across both the Lower and Upper Namoi catchments (Timms et al. 2009). Historic data was compiled and at-risk and representative bores across the region were re-sampled (Timms et al. 2009; Badenhop and Timms 2012). In that study, the variability of groundwater quality across the catchment and over time was augmented with new data. Standard protocols were used to test ~60 samples at 45 bores on three occasions during 2009 with a total of 189 field parameter records and 121 major ion analyses. Groundwater salinity was found to be relatively stable at most sites where sufficient historic data was available (105 monitoring pipes), although significant groundwater salinity increases occurred over the past two decades at about 20 % of sites. Salinity increases were most concentrated to the east and south-east of Gunnedah (Fig. 15.13). This is an area with intensive groundwater extraction and where naturally high salt stores within three metres of the surface occur (Fig. 15.13). One of the worst cases was a bore screened at 80 m depth where the average EC from monitoring in 2000–2009 (8.8 dS/m) was 156 % higher than the average from 1980 to 1999 (3.5 dS/m). In contrast, groundwater in other bores in the area was found to be fresh. An update to the original analysis (Badenhop and

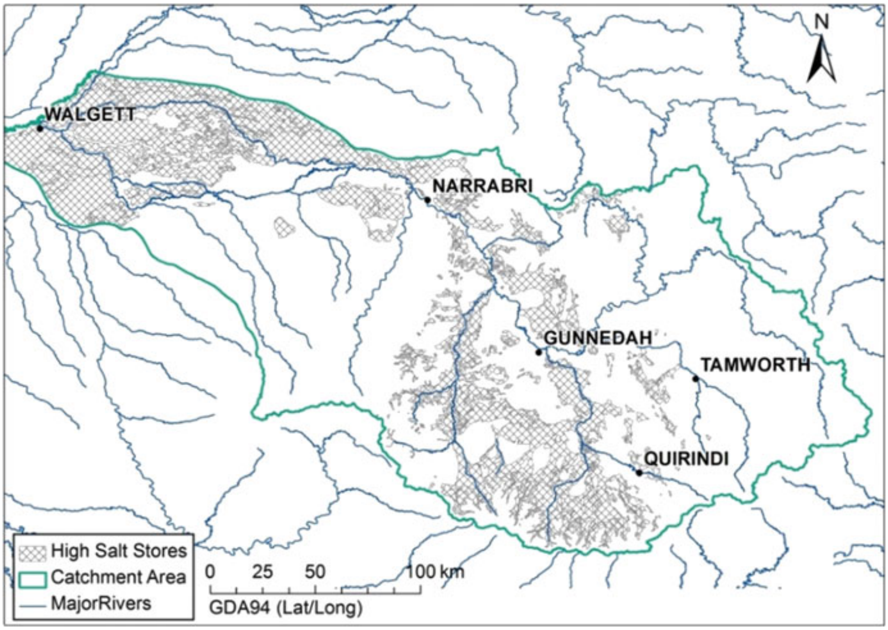

Fig. 15.13 Salt store in upper 3 m of soil in Namoi catchment (Namoi CMA)

Timms 2012) found that freshening had occurred at about 25 % of sites that had sufficient data over the same period (Fig. 15.14). This figure shows the lack of comparative EC data in many areas, and the spatial non-uniformity in salinization and freshening.

Earlier studies of groundwater salinity changes focused on processes at research sites or were limited in scope to restricted parts of the Namoi catchment. Agriculture-induced salinity is a well-documented soil and land management issue on the adjacent Liverpool Plains, a part of the Upper Namoi catchment (e.g. Ringrose-Voase et al. 2003) and broad trends of aquifer salinization since the 1980s have been identified by Lavitt (1999) in the Mooki River area, to the east of Gunnedah. These are attributed to extraction of groundwater and increased downwards flux of saline water from clayey silt deposits. In areas dominated by groundwater fed irrigation, complex hydrochemical variations occur to a significant depth in the system during and after significant periods of groundwater extraction (Timms and Acworth 2002; Timms and Ackworth 2009).

The potential for mixing of fresh and saline groundwaters induced by groundwater pumping is a major concern for both the Lower and Upper Namoi catchments. Studies by McLean et al. (2011) indicate that salinity is increasing at several hotspots, due to several factors including changing hydraulic gradients, and leaching of salt-laden sediments. The beneficial use of groundwater was found to have deteriorated since monitoring began, with some bores no longer suitable for

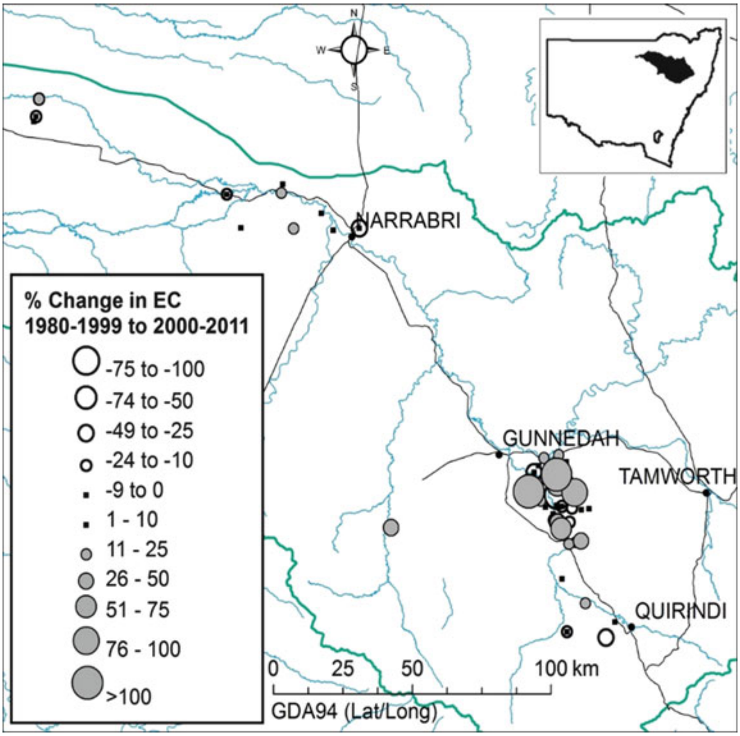

Fig. 15.14 Hot spot of groundwater salinity changes near Gunnedah showing a non-uniformity in salinization and freshening (After Badenhop and Timms 2012)

stock (generally noted in shallow bores), or no longer suitable for irrigation of some crops, including cotton (notably in the deeper bores).

Groundwater chemistry changes due to mixing induced by irrigation pumping were also observed in the Lower Namoi irrigation area by Barrett et al. (2006). The increase in salinity in this area, however, remained within the beneficial use limits for irrigation. In the western, arid parts of the Namoi, there is no irrigation to mobilize salts, but there is evidence (Timms et al. 2012) that clearing of native vegetation has caused a salt bulge in the soil to leach downwards. Total salt loads of 91–229 t/ha NaCl equivalent were measured for deep salt stores (from the ground surface to 10 m depth) for both perennial vegetation and cropping sites, despite salinity not being detected by shallow soil surveys, that are typically limited to 2–3 m depth. Groundwater salinity varied spatially from 0.9 dS/m to 2.4 dS/m at 21–37 m depth (N = 5), whereas deeper groundwater remained less saline (0.3 dS/m).

15.4.2.4 Salinization and Freshening Processes in Aquifers

Aquifer salinization in this inland groundwater system has been attributed to several processes. Multiple processes are required to explain the isolated and patchy nature of the temporal trends across the catchment. The most significant aquifer salinization processes in this area thus include: (i) leaching of saline soils with increased recharge; (ii) downward and lateral flow of shallow saline groundwater into deeper fresh aquifers as hydraulic gradients are enhanced by extraction of deep fresh alluvial aquifers for irrigation, and (iii) possible leakage that occurs at some locations via aging and poorly constructed bores whereby water from shallow saline aquifers leaks into deeper fresh aquifers.

All processes suggested by Foster and Chilton (2003) (Fig. 15.4) may have affected salinization of shallow groundwater in localised areas of the Namoi catchment. However, the processes that have led to salinization of deeper alluvial aquifers used for irrigation still require further investigation. Leaching of saline soils is likely to be more significant in areas with relatively high permeability soils and where ponding occurs in areas of relatively low surface elevation (Timms et al. 2012).

Shallow, saline groundwater has long been an issue for land management in the Namoi (Abbs and Littleboy 1998; Ringrose-Voase et al. 2003) and is also a threat to salinization of deeper fresh aquifers (Badenhop and Timms 2012). Mobilization of saline groundwater that occurs in stratified sediments, either via lateral flow, or from leakage from overlying sediments with variable salt content, could account for the patchy and localised nature of the groundwater salinity trends observed (Badenhop and Timms 2012). The degree of dilution of saline groundwater by fresh groundwater within aquifers is unknown, though mixing within pore waters of the layered sedimentary alluvium may be limited. Enhanced hydraulic gradients due to groundwater pumping for irrigation influence aquifers with good hydraulic connectivity, although there can be significant time delays for changes to occur is less hydraulically connected aquifers (Kelly et al. 2013).

Due to the possible leakage from aging, or poorly constructed, bores (Fig. 15.15) causing salinization of groundwater (Santi et al. 2006), newly revised construction guidelines (NUDLC 2012) for water bores in Australia emphasise the need for appropriate annulus seals, rehabilitation and decommissioning procedures (Timms and Acworth 2009). However, there is anecdotal evidence of irrigation bores having been recently constructed in the Namoi area with gravel pack filling the annulus between the deep alluvial aquifer to the surface, potentially allowing shallow saline groundwater a conduit for leakage into deeper fresh aquifers. For example, the salinity of fresh groundwater could double from 0.2 to 0.4 dS/m by mixing with just 1.7 % of saline groundwater at 12 dS/m. The National Water Commission estimated that NSW has a liability to replace at risk monitoring bores at a cost in the order of \$35.6 million, mainly for ~800 monitoring bores constructed of steel casing or screen that are aging (NWC 2012). Although many of the monitoring bore casings in the Namoi are PVC rather than steel that is susceptible to failure, no account has been taken of the failure of bentonite seals in the NWC (2012) review of groundwater monitoring infrastructure.

15.4.2.5 Freshening processes

Freshening of groundwater observed by Timms et al. (2009) at some bores near the river was attributed to changes in gaining and losing stream patterns. Many

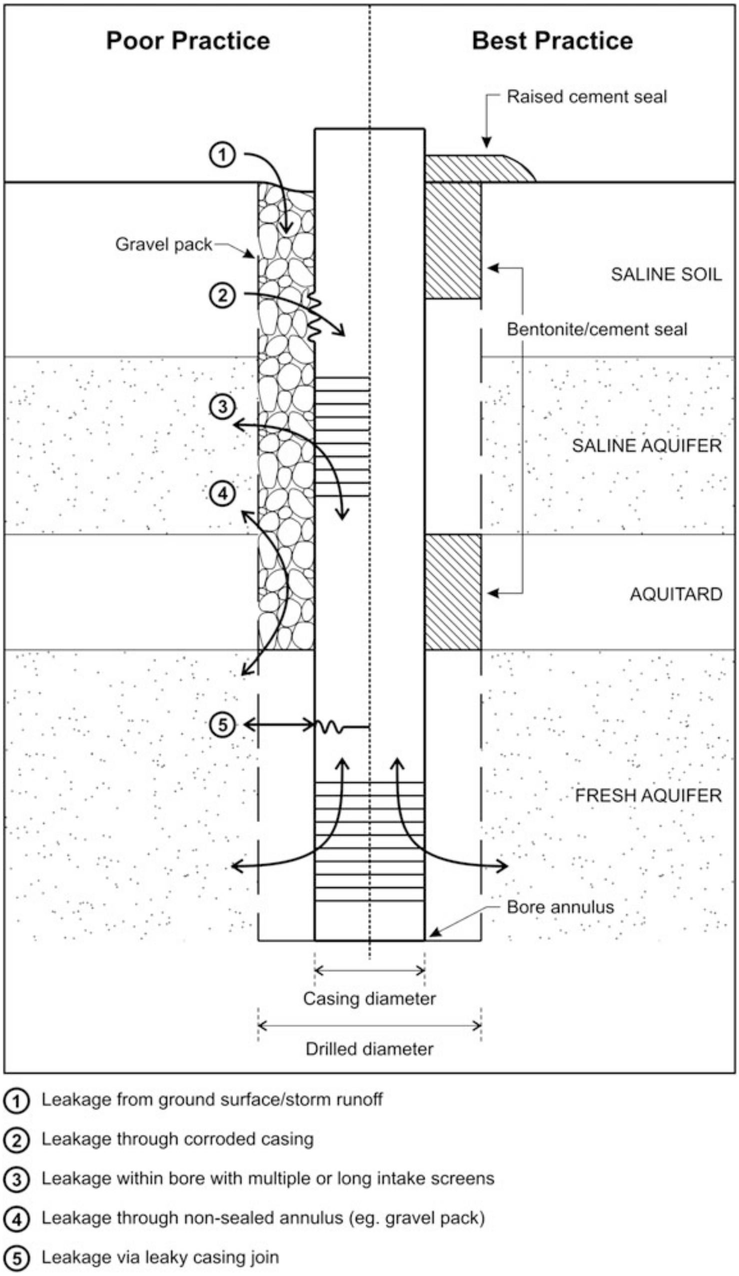

Fig. 15.15 Leakage pathways for groundwater in poorly constructed, or failed, groundwater bores (After Timms and Acworth 2009)

locations that were once gaining streams are now losing streams due to development of the groundwater resources (CSIRO 2007; Andersen and Acworth 2009; McCallum et al. 2014). Prior to extensive development of the Lower Namoi alluvium, recharge from stream losses would have been about 9 GL/yr., whilst from 1980 to 1998 stream loss accounted for an average of 41 GL/yr. This enhanced stream loss could lead to freshening of groundwater as river water is typically of lower salinity. The impacts of current levels of extraction on stream loss are yet to be fully realised (Kelly et al. 2013), and further changes in flow rates and salt fluxes will occur in the future.

15.4.2.6 Beneficial Use Impact

The NSW Groundwater Protection Policy contains the management principle that "All groundwater systems should be managed such that their most sensitive identified beneficial use (or environmental value) is maintained" (NSW DLWC 1998). Average EC for the period 2000–2005 was used by Timms et al. (2009) to assess the beneficial use across the catchment as a reference to determine if changes had occurred. The majority of waters across the Namoi Catchment were found to be suitable for drinking water, with waters only suitable for irrigation and livestock in selected areas of the Upper Namoi, and in the north-west as well as those areas furthest from the Namoi River in the Lower Namoi alluvium.

An analysis was completed by Timms et al. (2009) to determine if there had been any recent changes in beneficial use, comparing data from 2000 to 2005 to data from 2006 to 2009. Unfortunately, only 27 of 1,268 monitoring bores had sufficient data for both time periods. Of these bores, only one showed a degraded beneficial use category within the broad definitions of drinking, irrigation and poultry, and livestock (Table 15.2). However, if the resolution of beneficial use category is increased to define changes that inhibit the growth of specific crops, the findings change. While mature cotton can be irrigated with water EC up to 7.7 dS/m, early season cotton tolerates only water with EC $<$ 1.5 dS/m (Fig. 15.16). A risk assessment of groundwater resources in the Namoi identified four areas where changes in salinity might occur in the future that require strategic monitoring and management strategies.

15.4.2.7 Integrated Management Responses

Management responses to groundwater quality issues include: local initiatives; regional efforts to understand and raise awareness, and state-based regulatory approaches. These approaches include policy and compliance functions consistent with Murray-Darling Basin Authority and National water reforms (Holley and Sinclair 2012 and Chap. 9).

Local Level

At a local level, the keys to efficient on-farm irrigation water management are knowing how much water in the soil profile is available to the crop and how much water the crop needs (Charlesworth 2005). This will minimise accessions of surface water and potential salt leaching into lower fresh aquifers.

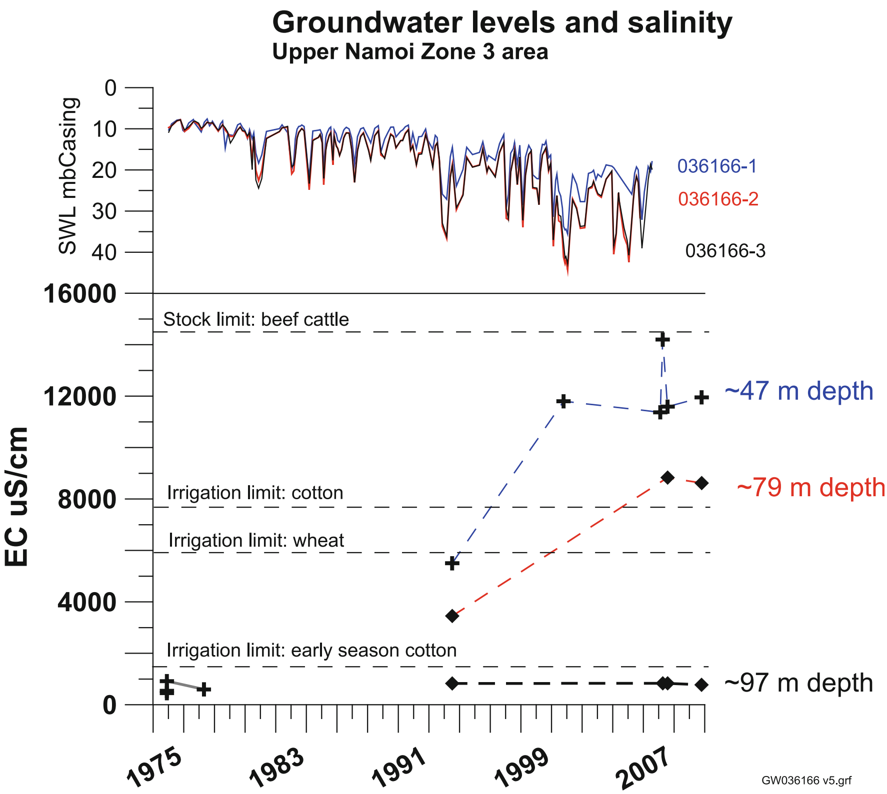

Fig. 15.16 Groundwater level and salinity changes over time compared to beneficial use guidelines at site GW036166 (After Timms et al. 2009)

Measuring and monitoring soil water status should be essential parts of an integrated management program. Irrigation managers and the irrigation service sector now have a large range of equipment available for measuring the soil water status, with favourable cost-benefits, water savings and crop yields. The Cotton industry has developed best management practices for irrigation (CRDC 2008) for irrigators, which includes a protocol for monitoring the salinity of bore water (Timms et al. 2009). Recommendations for management responses include calls for more strategic monitoring, improving data integrity and archiving, understanding of processes and numerical modelling of aquifer salinity changes that could occur in the future (Timms et al. 2009; McLean et al. 2011; Kelly et al. 2013).

Assessing and managing irrigation salinity on a farm by farm basis includes a range of possible investigations, including soil type and EM surveys, water salinity and depth to water table monitoring. Best practice irrigation requires scheduling to match plant water requirements at differing stages of the season and taking into account soil type and salinity. Excess drainage is to be avoided, or reduced, in salt prone areas, or managed with more efficient irrigation methods such as drip and spray application.

Strategic groundwater monitoring guidelines were developed by Timms et al. (2009) with a 4 level Best Management Practice (BMP) for irrigation bores, and a 3 level guideline for sub-catchment and regional scale. For example, a level 2 BMP for irrigation bores is to maximize crop yields by using bore water within appropriate salinity guidelines.

Regional Level

At a regional level, the existing State Government Water Sharing Plan has set a sustainable yield (also referred to as the diversion limit) of 86,000 ML per year (NSW 2008; Smithson 2009). Kelly et al. (2013) demonstrated (at the 90 % confidence interval) that under this rate of withdrawal the groundwater level will continue to fall, which is to be expected given that groundwater hydrographs indicate that dynamic equilibrium has not been reached. The sustainable-yield groundwater flow modelling undertaken by CSIRO (2007) indicates that under some climatic scenarios dynamic equilibrium will not be reached within at least 100 years. Thus, the current Namoi Catchment Action Plan 2010–2020 (NCAP 2010) goal of not allowing the groundwater levels to fall cannot be achieved without reducing groundwater withdrawals, or changing the way both surface-water and groundwater are distributed and used throughout the whole of the Namoi Catchment. Kelly et al. (2013) argue that if groundwater is allocated and managed only in the context of point of use, or in assumed isolation from surface water, sustainable access to groundwater for all existing irrigation farms will be difficult to attain while minimising the impact on groundwater-dependent ecosystems. This will only be achieved if surface-water and groundwater are managed as a single resource at the catchment scale.

State Level

The NSW State Groundwater Policy Framework (NSW DLWC 1997) highlights the need to manage the access to groundwater within the sustainable yield of a system so that the availability of the resource is sustained for all consumptive uses as well as the dependent ecological processes. Groundwater quantity, groundwater quality (NSW DLWC 1998) and groundwater dependent ecosystems are specifically addressed in the framework. Status reports are regularly prepared on the basis of monitoring in groundwater management areas and zones. Local groundwater allocations and trading rules are decided on the basis of groundwater level and quality status assessments. Emphasis is placed on addressing hotspot issues and on salinity management in areas where downwards leakage of saline water is occurring due to extraction. For example, Smithson (2009) used saturated thickness depletion limits and water level constraints to define local impact management rules in the Lower Namoi alluvium.

In 2012, the Aquifer Interference Policy (AIP) was introduced to clarify rules for groundwater licencing, with particular implications for coal bed methane and mining projects (NSW 2012). The AIP covers activities that interfere with an aquifer including penetration, obstructing flow, taking or disposing of water. The AIP also includes considerations for dewatering for infrastructure and injection of waste into groundwater systems for both individual and cumulative impacts. The assessment criteria are called 'minimal impact considerations' and include impacts on water table levels, water pressure levels and water quality in different types of groundwater systems.

The state Office of Water (NOW), a stand-alone regulator, has responsibility for regulation under the *Water Act 1912* (NSW) which is gradually being superseded by the *Water Management Act 2000 (NSW)* (WMA 2000). Holley and Sinclair (2012) reviewed compliance and enforcement of water licences in NSW and deduced that over 24,000 surface and groundwater licences were covered under the *WMA*, accounting for 95 % of all water extracted in NSW. The *WMA* 2000 details a range of breaches with potential application to groundwater, including taking water that is not authorised by a licence, constructing a bore without approval, not maintaining a water meter and interfering with an aquifer.

15.5 Integration and Conclusions

Given the increased dependence of groundwater globally to maintain societies and their economies, it is critical to combat the various types of degradation threatening groundwater supplies. This chapter has looked at the issue of groundwater salinization, one of the major forms of aquifer degradation, from a theoretical and practical view.

Assessment of the physical and chemical processes causing salinization of the root-zone and shallow aquifers identified three major types of salinity based on soil and groundwater processes: (i) groundwater associated salinity, (ii) transient salinity and (iii) irrigation induced salinity (Rengasamy 2006). These types of salinity all result in accumulation salt stores in the root zone which can have major effects on plant growth and soil processes. The salt stores can also affect groundwater supplies deeper in the landscape. Three processes were specifically recognised as causing salinization of aquifers: (i) rising groundwater tables due to inefficient surface irrigation and inadequate drainage, (ii) natural salinity mobilized from the landscape due to land clearing of native vegetation, and (iii) disturbance of natural groundwater salinity stratification by well construction and groundwater extraction (Foster and Chilton 2003).

The extent of the salinization problem, including the influence of management and policy, was explored using two case studies representative of a developing country (Pakistan) and developed country (Australia).

The Indus Valley (Pakistan) case study highlighted deep percolation from the irrigation system of canals and lack of drainage that resulted in a major imbalance in the amount of salt entering and leaving the soils and hence salt accumulation in the shallow soils. A cost benefit analysis of implementing three management strategies (no intervention, an engineering approach, and an agronomic approach) on four land categories was carried out. The four land categories ranged from extremely degraded land through to normal land and were based on the severity of waterlogging (depth of water table) and the soil salinity. The Net Present Values (NPV) of the three strategies on the four land categories were compared over a period of 25 years, using a discount rate of 6 %. The results indicated that some strategies can maintain farm profitability in the long run, provided that the right kind of treatment is given to the right type of land at the right time.

The Namoi catchment (Australia) study demonstrated that aquifer salinization occurs from salt remobilisation in the landscape, with redistribution by changing groundwater flows. The source of salt for shallow aquifers that are located inland, far from possible sea-water intrusion, includes salt naturally accumulating in the soil, or leakage from saline aquifers located overlying, adjacent to, or below freshwater aquifers. A detailed monitoring program indicated that both salinization and freshening processes occurred in aquifers at different depths. The most significant aquifer salinization processes in this area include: (i) leaching of saline soils with increased recharge; (ii) downwards and lateral flow of shallow saline groundwater into deeper fresh aquifers as hydraulic gradients are enhanced by extraction of deep fresh alluvial aquifers for irrigation, and (iii) possible leakage that occurs at some locations via aging and poorly constructed bores.

Both the case studies from the Indus Valley and the Namoi Catchment involved examples of how management strategies and policy can be integrated to reduce salinization of the root zone and aquifers respectively. Thus:

(i) The Indus Valley case study demonstrated how it is now feasible for planners to make informed decisions on the rehabilitation of salinized surface soil. Previously, investments in salinity had not provided economic returns. Therefore, despite great interest, funding agencies had little confidence in making investments for salinity management. The approach taken in this case study linked viable policy management that has economic credibility to well-documented, and practical, salinity management.
(ii) In the Namoi case study, management responses to groundwater quality issues require both local initiatives and regional efforts to understand and raise awareness and inform state-based regulatory approaches. These approaches include policy and compliance functions consistent with Murray-Darling Basin Authority and National water reforms (Holley and Sinclair 2012).

In summary this chapter on root zone and groundwater salinization processes has indicated that as the surface root zone can be a major source of salts that lead to aquifer salinization, it is critical that both the interactions within and between these zones (i.e. root zone and aquifers) are understood if salinization of valuable groundwater supplies is to be prevented.

It is important to note that this chapter has focussed on the biophysical processes in the root zone and aquifers and their interactions that are likely to cause groundwater salinization. As demonstrated by the two case studies, management strategies to reduce salinity both in the root zone and/or in aquifers, also need to be carefully integrated with policy. Such integration needs to consider the following:

(i) Land use policies to reduce salinity need to take into account the suitability of the soil and the type of crop;
(ii) The nature of local and regional water allocation policies need to be integrated;

(iii) Agricultural and agronomic policies that encourage groundwater use in salinity affected areas can accelerate the processes of aquifer salinization.
(iv) There is a need to understand and closely monitor catchment scale processes and their feedback on each other for sustainability of land and water resources. Related to land and aquifer salinization such catchment scale processes may include the clearing of deep rooted native vegetation for agricultural extension, over extraction of groundwater, and the use of saline and sodic water for irrigation. The feedbacks from each of these processes can accelerate land and aquifer salinization.
(v) It is important not to degrade the resources (land and water) in the first place as some processes such as soil salinization and aquifer degradation can be non-reversible.
(vi) Integrated catchment management is urgently required.

Other chapters in this book deal in more detail with the issue of how to integrate management strategies to reduce salinity with policy, e.g. Chap. 2.

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Managed Aquifer Recharge: An Overview of Issues and Options

16

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Abstract

As covered in Chap. 2, many of the world's aquifers are rapidly being depleted. Nearly one quarter of the world's population – 1.7 billion people – live in regions where more water is being consumed than nature can renew (Gleeson et al. 2012). Over-exploitation occurs when groundwater abstraction is too intensive, for example for irrigation or for direct industrial water-supply like extracting fossil fuels (Pettenati et al. 2013; Foster et al. 2013). When groundwater is continuously over-pumped, year after year, the volume withdrawn from the aquifer cannot be replaced by recharge. Eventually, the groundwater level is much lower than its initial level and even when pumping stops, the aquifer has trouble rising once again to its original level. In continental zones, over-exploitation can lead to groundwater drawdown and, ultimately, to subsidence through development of sinkholes when underground caverns or channels collapse. In coastal areas, the decrease in groundwater recharge results in saltwater intrusion into the aquifer formation (Petalas and Lambrakis 2006; De Montety et al. 2008). Preserving local groundwater resources is an environmental and economic issue in coastal zones and is vital in an island context. The increasing demand for water caused by a growing population can lead to the salinization of groundwater resources if these are systematically over-exploited. Limiting the salinization of coastal aquifers is consistent with the groundwater objective of the European Union Water Framework Directive, which is to achieve a good qualitative and quantitative status by 2015. The economic advantage of preserving these threatened water resources is that, when there is a growing demand, a local water resource is sustained and there is no need to import water. Transporting water can cost 2–10 times more than limiting the intrusion of saltwater into a coastal aquifer.

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16.1 Introduction

As covered in Chap. 2, many of the world's aquifers are rapidly being depleted. Nearly one quarter of the world's population – 1.7 billion people – live in regions where more water is being consumed than nature can renew (Gleeson et al. 2012). Over-exploitation occurs when groundwater abstraction is too intensive, for example for irrigation or for direct industrial water-supply like extracting fossil fuels (Pettenati et al. 2013; Foster et al. 2013). When groundwater is continuously over-pumped, year after year, the volume withdrawn from the aquifer cannot be replaced by recharge. Eventually, the groundwater level is much lower than its initial level and even when pumping stops, the aquifer has trouble rising once again to its original level. In continental zones, over-exploitation can lead to groundwater drawdown and, ultimately, to subsidence through development of sinkholes when underground caverns or channels collapse. In coastal areas, the decrease in groundwater recharge results in saltwater intrusion into the aquifer formation (Petalas and Lambrakis 2006; De Montety et al. 2008). Preserving local groundwater resources is an environmental and economic issue in coastal zones and is vital in an island context. The increasing demand for water caused by a growing population can lead to the salinization of groundwater resources if these are systematically over-exploited. Limiting the salinization of coastal aquifers is consistent with the groundwater objective of the European Union Water Framework Directive, which is to achieve a good qualitative and quantitative status by 2015. The economic advantage of preserving these threatened water resources is that, when there is a growing demand, a local water resource is sustained and there is no need to import water. Transporting water can cost 2–10 times more than limiting the intrusion of saltwater into a coastal aquifer.

All over the world, the problems related to groundwater withdrawal from coastal aquifers are usually complicated because they associate the notion of quantity with that of quality (Werner et al. 2013). They are even more complex given that there is often a high demand for water in coastal areas. Population growth and the development of agriculture, industry and tourism are leading to increased groundwater abstraction, while the effects of global climate change are increasing seasonal variations. The deterioration of groundwater quality and quantity as demand increases is becoming more pronounced. In order to prevent this salinization, long-term operational management measures must be taken.

In such various contexts, technologies for Managed Recharge Aquifer (MAR) are of particular interest (see also Chap. 17). Indeed, these technologies aim to increase the available quantities of groundwater by increasing groundwater infiltration to aquifer formations. Together with rain, treated wastewater and desalinated seawater, it is one of the unconventional sources of water that is most often included in integrated water management schemes. MAR is one of the measures that can be implemented to secure water supply, compensate for some effects of climate change and, more generally, handle the quantity and quality of groundwater bodies. It is not, however, a substitute for groundwater management based on decreasing abstraction and adapting withdrawal to resource availability.

Due to its relatively high cost (Khan et al. 2008), MAR has been practised, over the last four decades, mainly in developed countries. It is commonly carried out in the United States, and now increasingly in Europe (Levantesi et al. 2010). Large cities, especially, most often use this management method (Berlin, Paris suburbs, Lyon, Dunkirk, Geneva). In these cities, MAR is used to manage stormwater by collecting surface runoff waters in infiltration basins. However, stormwater can be one of the main sources of pollutants (heavy metals, hydrocarbons and other organic compounds) produced by cities. Consequently, traditional urban drainage systems now cause many technical and environmental problems, notably the pollution of the surface receptor media (Chocat et al. 2007). Nevertheless, some MAR technologies can also be used to limit the pollution of surface water by infiltrating some of the polluted water and monitoring the geo-purification and/or attenuation processes. Therefore, MAR can also be undertaken to protect the environment by limiting the level of pollution in sensitive receptor media. In Mediterranean countries, MAR has in some cases been taken into account in reservoir design in order to limit losses by evaporation. In coastal contexts, the MAR objective is to move from passive management of saltwater intrusion (by reducing abstraction for the drinking water supply) to dynamic management – optimising pumping and natural and artificial recharge as a function of aquifer model predictions and the results of continuous, in-situ monitoring of the water table. The principal objective of MAR here is to create a hydraulic barrier to prevent the intrusion of pollutants and saltwater (Casanova et al. 2007, 2008).

Based on the large number of existing reviews already published about MAR technologies (e.g. Dillon et al. 2009a, b; Page et al. 2010; Chap. 17), the present book chapter summarizes the main managed recharge measures (types of artificial recharge systems, hydrogeological and regulatory restrictions, health and environmental risks) and makes recommendations concerning site selection, technical feasibility and monitoring methods. As an illustrative purpose of MAR technologies, there is a focus on French MAR installations.

16.2 MAR Technologies

16.2.1 Infiltration Methods

Infiltration methods are designed to facilitate the infiltration of water to the aquifer by means of infiltration basins (ponds, tanks), while improving the quality of the recharge water by natural attenuation in the aquifer's unsaturated zone. They are usually used to recharge water table aquifers or, in some cases, to create hydraulic barriers. One of the main advantages of these methods is that they are inexpensive and relatively easy to implement and maintain. The "infiltration ponds" method includes installations such as dams and small reservoirs, as well as various improvements and installations designed to manage stormwater (or runoff) and individual wastewater treatment units. This category also includes reservoir pavements, recharge pits, drainage trenches, vegetated ditches, mounds systems, sand filters and septic drain fields.

With all of these techniques, the water passes through the unsaturated zone before reaching the aquifer. The water can thus be potentially purified by contact with the soil, which enables the elimination not only of certain pathogenic agents but also of potentially harmful inorganic and organic substances. Infiltration ponds are often used in MAR projects, notably in places where there are frequent water shortages. Depending on which technique is being considered (infiltration ponds, percolation tanks, Soil Aquifer Treatment (SAT) Dillon 2005), basin characteristics such as the desired infiltration rate are adapted to the local objectives and can vary significantly. If the objective is quantitative, the chosen infiltration rate might be relatively high (several meters per day), whereas a lower infiltration rate (an average of 0.5 $m.day^{-1}$) would be recommended when the objective also includes the geo-purification of the infiltrating water.

The geo-purification capacity of the filtering layer is in some cases improved when plants are grown on this layer. Indeed, the presence of plants in the filtering layer protects the surface of the basin from erosion and clogging and is carriers for bacteria that act for biodegradation of some inorganic and organic pollutants. On summer, plants can improve the purification of the infiltrated water by enhancing phytoremediation.

The water temperature also has an influence on the infiltration rate. Colder water infiltrates more slowly due to an increase in viscosity. The volume of water that infiltrates below a basin can therefore decrease significantly in winter. Water that has not been greatly treated is also rich in organic matter, which fosters the development of bacteria. This might decrease the porosity, notably by the formation of biofilms.

To avoid, or rather slow down this clogging process several processes are available, depending on the application (Le Coustumer 2008). The first is to pre-treat the water that will infiltrate. Settling ponds or sand filters can be installed upstream from the infiltration basin, or the chemical properties of the recharge water can be modified by adding various chemical compounds, mostly inorganic. The second solution is to operate infiltration basins alternately, following “wetting-drying” cycles, in order to enable the decompacting and/or maintenance of the basin floor.

16.2.2 Direct Injection Method

Direct injection methods using injection wells are the methods most often used throughout the world. Aquifer Storage and Recovery (ASR) and Aquifer Storage Transfer and Recovery (ASTR) are installed mainly to meet two objectives: (i) to recharge confined (or semi-confined) aquifers and/or (ii) to create hydraulic barriers. The quality of the injected water must be closely monitored to prevent any contamination. They might also be preferred when space is limited because their footprint, only several tens of m^2, is small compared to that of infiltration basins. Moreover, their wellhead protection zone is small.

The principle behind ASR is the injection of water into an aquifer followed by its recovery by pumping from the same well at a later date. This method involves distinct and alternating periods of storage of excess water and of its consumption (Pyne 2006). ASR is therefore generally used for aquifers that are relatively invulnerable to non-point source pollution and in which groundwater moves slowly, i.e. confined or semi-confined aquifers. This method is used mainly for the seasonal storage of good quality water (sometimes potable), as a "pocket of fresh water" in an aquifer containing non-potable water. From a technical point of view, there are two advantages to this method. First of all, it entails alternating phases of injection and abstraction in the same well. This results in an inversion of the water circulation in the well screen and in the surrounding aquifer, thereby reducing clogging (Dillon et al. 2006; Pyne 2005, 2006). Secondly, the use of the same well for injection reduces investment costs.

As opposed to ASR, ASTR involves injection in one well and recovery by pumping from a second well located several hundred metres down-gradient from the injection well. The injected water is transferred through the aquifer before being abstracted. The specific technical characteristics of this set up require that the water in the aquifer be of relatively good quality. ASTR is therefore used mainly when the pumped water is a supply for human consumption.

16.2.3 Filtration Methods

Induced recharge called riverbank filtration, involves increasing the infiltration of water from a river to its alluvial aquifer by pumping in wells located near the riverbank. A string of wells are installed parallel to and near the river. Pumping in the wells lowers the water table, creating a difference in head between the river and the groundwater. This draws the surface water through the riverbank, as long as the riverbank is not clogged and/or the pumping rate is sufficient. The principal aim of this technology is to use the geo-purifying capacity of the riverbank to filter and purify the recharge water. Due to the high concentration of suspended matter in surface water, riverbanks rapidly become clogged. In order to prevent this, the infiltration rate must be relatively low and the riverbanks must be periodically maintained. Another method called dune filtration consists of infiltrating water from ponds constructed in dunes. The water is then extracted from wells or ponds at lower elevation (Dillon 2005).

16.3 Sources of Water Used for MAR

The first basic criterion concerning the feasibility of a MAR project is the availability of recharge water near the injection site in order to ensure a steady supply and limit potential transport costs. An aquifer can be recharged with several types of water. Several types of water are used for recharge: surface water from rivers, stormwater and treated wastewater.

The inventory of MAR installations still active in France showed that almost all of these use surface water, notably due to the availability of this resource. Indeed, surface water is abundant in temperate countries where rainfall adequately replaces water lost by evapotranspiration and flow to the sea. There are three other reasons for the predominant use of surface water. Firstly, the chemical and microbiological quality of this water is adequate, even when the water is not pre-treated, which enables its use for both quantitative and/or qualitative objectives. Secondly, surface water can be used with different existing MAR techniques, from infiltration or indirect injection to direct injection. Thirdly, the laws that enable the use of surface water for recharge systems already exist.

Intermittent surface water bodies can also be used, although their hydrological cycle is somewhat random and water availability depends on climate events that only occur over several days or weeks each year. It is important to note that the duration of these climate events can vary from year to year. Historically, this type of surface water has been little used for MAR due to its intermittent character. Recently, new techniques, notably for direct injection, have been developed in order to use this type of water. Most of these are new ASR techniques developed in semi-arid and arid Mediterranean climate zones.

Desalinated water made from seawater or brine is an alternative. First developed to produce drinking water, industrial water and water for agriculture, desalinated water can also be used for MAR. Initially used only in energy-rich countries like Saudi Arabia, the United Arab Emirates or Bahrain (Ahmed et al. 2001; Al-Zubari 2003; Chafidz et al. 2014), all of which produce drinking water from seawater, desalinated water is increasingly used elsewhere in the world thanks to improved desalination techniques that have decreased production costs (Shatat et al. 2013; Feitelson and Rosenthal 2012; Moatty 2001; Palomar and Losada 2010). Because of the relatively small quantities produced and their very high cost (Dabbagh 2001), desalinated water is almost never used for MAR, the aim of which is to significantly increase the volume of groundwater. However, the stability of desalinated water production might be a favourable argument for its use in some arid countries as a secondary source of recharge water for installations whose objective is quantitative. Desalination techniques confer particular chemical properties on this type of water. The principal characteristic is that it contains very little salt. When water is produced by distillation, it usually has a dissolved salt content of between 5 and 30 mg.L^{-1}. Due to its low salt content, this water does not meet drinking water standards. It is therefore necessary to remineralize it until its salt content reaches ca. 300 mg.L^{-1}.

At present, treated wastewater is used in MAR systems in many countries. Although very common in countries with limited water resources, wastewater is rarely reused in France (about 40 projects developed experimentally for irrigating crops, watering golf courses and forests or prairies) and there are no MAR installations that specifically use treated urban wastewater. It is important to point out that in France the use of wastewater for MAR is forbidden (Miquel 2003).

The volumes of reclaimed treated wastewater are slightly higher than those of desalinated water but are still lower than those of surface water. Like that of desalinated water, the production of treated wastewater is relatively stable over time. It is, however, important to point out that the production of wastewater increases drastically during tourist seasons in holiday resorts. Therefore, treated wastewater is usually used for both recharge systems whose objective is to significantly increase the volume of groundwater and those whose objective is to improve groundwater quality.

In this context, MAR using infiltration of treated wastewater might be one of the possible solutions for recycling water to its natural medium while making it possible, for example, to recharge over-exploited aquifers, prevent saltwater intrusion into coastal aquifers, or store water without the loss by evaporation that occurs in open-air reservoirs, and make it available during periods of high demand. Treated wastewater is therefore an alternative resource that is available throughout the year and, in particular, during low water stages when the demand for conventional resources is highest, or when they are unavailable. It is of particular interest when the natural resource is scarce, notably in coastal areas and on islands. Moreover, the infiltration of treated wastewater through an unsaturated zone to recharge an aquifer benefits from the purifying capacity of the sub-surface in which naturally occurring processes enable the degradation or filtering of a certain number of the water's pollutants (Bekele et al. 2011).

Industrial water comes from factories, manufacturing plants and farms. The discharge of this water is subject to a specific study and preliminary treatment is usually required. It can contain both easily degradable organic compounds and substances that do not degrade easily such as organohalogenated compounds or heavy metals. Compared to treated urban wastewater, industrial water which contains more specific contaminants (organic molecules, trace metals and contaminants of emerging concern (CECs)) is generally not used for MAR. Indeed, most companies have their own treatment plants to treat the specific effluents of their industrial processes. This means that this water is not systematically discharged to the municipal wastewater system, thus limiting its use. However, this water can be used for MAR if there is a system to route the water to the MAR site and its chemical quality has been specifically studied.

16.4 Hydrogeological and Regulatory Constraints

The feasibility of an MAR system depends for the most part on local hydrogeological conditions (Dillon 2005). In the case of infiltration methods, the unsaturated zone must allow the water to infiltrate to the aquifer and the aquifer must be able to store the infiltrated water. Preference is therefore given to sites that have a rather low diffusivity, i.e. relatively low permeability and high storage capacity. These conditions can be found in aquifer formations with interstitial porosity (e.g. sandy, sandstone formations) or with both interstitial and fracture porosity (e.g. chalk). As concerns water quality, when choosing an MAR site, one must be sure that the quality of the recharge water is compatible with the reactive potential of the aquifer matrix and especially that of the unsaturated zone.

Current French regulations state that an MAR system is subject to prior approval in compliance with the environmental code and an impact assessment must be carried out. It must comply with French and European water laws, in particular with respect to the prevention and mitigation of discharge of pollutants to groundwater. In the specific case of MAR systems, the environmental code prohibits the use of treated wastewater in France.

Wells used to supply drinking water can be located down-gradient of the sector targeted for MAR with water whose quality is degraded. It is therefore essential that the safety for public health and the environment of the artificial recharge, induced by the addition of water to a parcel and its transport to the aquifer through the unsaturated zone, is ensured. The regulatory "wellhead protection zone" tool, described in the French public health code (Water law of the 3th January 1992, article L-1321-2) is, in most hydrogeological contexts, poorly suited to preventing pollution. Additional measures have therefore been taken in protection zones. These must now be implemented at the scale of an entire well or well-field catchment area, which is the most appropriate spatial unit for combating non-point source pollution (Vernoux et al. 2010). If this catchment area includes an MAR installation, the restrictions on the quality of the infiltrated water are even stricter.

16.5 Health and Environmental Risks

Depending on the quality and the efficiency of the treatments given to the recharge water, it can contain various amounts of pollutants such as trace metals, nutrients and microorganisms, including pathogenic microorganisms and contaminants of emerging concern (CECs) (Lapworth et al. 2012). Using different waters that have different origins and different qualities, notably treated wastewater, for MAR systems might therefore create high risks for public health. The complexity of reactive transport processes in the unsaturated zone highlights two of the main stumbling blocks that must be taken into consideration if treated wastewater is being considered for MAR: one specific challenge is to have numerical models that can include all of the hydro-biogeochemical processes involved in reactive transport, while a second, more operational, is the need to have a complete biogeochemical and hydrogeological characterisation specific to each MAR site.

16.5.1 Trace Metals

The problem posed by metals in recharge water concerns first of all the use of treated wastewater since the concentrations of many trace metals are very low in most natural waters. Several studies have shown that trace metal concentrations can vary greatly in runoff and surface water but that, except for iron and lead, they are, the most part of the time, below acceptable levels (Haeber and Waller 1987). Iron and lead present relatively few health risks. Recharge water coming from water treatment plants might also contain trace metals, the most abundant of which are iron, zinc, copper and lead. Other trace metals can also be found: manganese, aluminium, chrome, arsenic, selenium, mercury, cadmium, molybdenum, nickel, etc. They are of various origins. They come from products consumed by the population at large, from the corrosion of material in the water distribution and treatment systems, from service activities (health, automobile) and possibly from industrial effluents (Cauchi et al. 1996). Trace metals can be dissolved in recharge water from the aquifer material by modification of natural geochemical conditions. The recharge water, rich in nutrients and organic matter leads to the creation of new redox conditions in the system driven by the microbial community (Hunter et al. 1998; Kloppmann et al. 2012; Pettenati et al. 2012).

Several countries (the United States and Australia, for example) have developed guidelines for the use of treated wastewater for recharge (USEPA 2004, 2012; WHO 2006a, b). These guidelines focus mainly on the health and environmental risks that result from the presence of pathogenic microorganisms, suspended solids and dissolved organic carbon in this water. There are few recommendations concerning trace element contents in water (e.g. USEPA 2012), except as concerns five trace metals. These are: (i) arsenic, for which the drinking water limit is 10 μg/L in France; (ii) nickel, which is only weakly toxic but which accumulates in plants; (iii) cadmium, which is considered to be the metallic pollutant of greatest concern due to its rapid accumulation in plants and its proven toxicity even at low concentrations (acceptable daily intake (ADI) 0.057 mg/day/individual); (iv) mercury, which can be highly mobile; and (v) lead, the injection of which, even at low doses, can cause neurotoxic and hepatotoxic disturbances (Dillon et al. 2009a).

16.5.2 Emerging Pollutants

Water quality and societal wellbeing are currently threatened by emerging pollutants and pathogens including antibiotic resistant bacteria and viruses. The recharge water that is most likely to be contaminated by pharmaceutical products is treated wastewater. Indeed, there are several sources of pharmaceutical products discharged to water bodies. The excretion of pharmaceutical products by patients following their ingestion is the main source of wastewater contamination. Hospital wastewater therefore contains high levels of pharmaceutical products, essentially antibiotics. Moreover, anaesthesia products, disinfectants and diagnostic products are also present in this wastewater.

Some pharmaceutical products in their active forms, and/or their metabolites if these are also active, can be hazardous for the environment from an eco-toxicological point of view. They are then found in wastewater. When wastewater is treated, the elimination of these pharmaceutical products and/or their metabolites varies depending on both the nature of the drug under consideration and on the characteristics of the treatment methods used in the treatment plant (Joss et al. 2005; Yu et al. 2006). Furthermore, the elimination of pharmaceutical products does not mean their total destruction. They can degrade into products that are also active (Kümmerer et al. 1997; Zwiener et al. 2002). Several studies have identified the presence of various pharmaceutical products in treated wastewater (Steger-Hartmann et al. 1996; Kümmerer et al. 1997; Ternes et al. 1998). For example, Ternes et al. (1998) showed that the most abundant pharmaceutical products in wastewater are beta blockers, contrast media and pain relief/anti-inflammatory drugs.

Like that of trace metals, the mobility of pharmaceutical products can be reduced with infiltration basins and indirect injection methods because the presence of an unsaturated zone enhances the trapping of these pollutants. In the unsaturated zone, geochemical and microbiological processes can indeed decrease the concentration of both pathogenic and non-pathogenic microorganisms and CECs by (i) biodegradation and (ii) adsorption. To a lesser extent, adsorption can also limit the mobility of organic pollutants. Pharmaceutical products can be adsorbed on several solid phases in the unsaturated zone, such as oxyhydroxides, mineralogical clays and humic substances. This adsorption of pharmaceutical products requires the creation of a chemical or electrostatic link between the functional groups present on a pharmaceutical product and the functional groups present on the solid phases in the unsaturated zone. The adsorption of these pharmaceutical products might or might not result in the release of chemical compounds to the aqueous phase.

16.5.3 Risk Assessment

At present, most studies have focused mainly on notions of environmental risk (Devaux 1999; Wintgens et al. 2012; Dillon et al. 2009b). These risk assessment studies consider three types of risks: (i) potential theoretical risk, (ii) potential experimental risk, and (iii) real risk.

Potential theoretical risk is related to all of the disruptions that might affect the various characteristics of the aquifer as a result of the installation of an MAR system. These include, for example, groundwater contamination by recharge water that contains pollutants (trace metals, metalloids, microorganisms, pharmaceutical products, etc.). The assessment of theoretical risk therefore requires the determination of: (i) the possible sources of contamination of the recharge water used, such as prolonged contact with minerals rich in trace metals, industrial discharge, or the presence of a nearby hospital; and (ii) the intrinsic chemical and microbiological quality of the recharge water.

Potential experimental risk corresponds to the risk that the disruptions affecting the recharged aquifers might be transferred to humans or to the environment. This experimental risk corresponds, for example, to the probability that a pollutant present in the recharge water will reach humans. In this case, the experimental risk will depend not only on the theoretical risk associated with the contamination of the recharge water but also to other factors such as the volumes of recharge water injected, the efficiency of pre-treatments, and the geo-purification capacity of the unsaturated zone in the case of infiltration structures (infiltration basin and indirect injection techniques).

The last type of risk that is considered by studies assessing the impacts of recharge systems is the real risk, which is the probability that one member of an exposed population will be contaminated (Devaux 1999). This risk broadens the notion of potential environmental risks by considering other factors that are specific to individuals exposed to disruptions caused by MAR such as the specific immune-system capacity of a given individual (natural or acquired), age, sex, health, nutrition, hygiene and the diagnostic ability of health personnel (e.g. serology).

Although many risk assessment studies have made it possible to define the conceptual framework of the risks associated with MAR, the complexity of the developed markers, and the lack of knowledge concerning some of the components of these markers, means that the dangers associated with the disruptions caused by this activity are hard to quantify. For example, the water consumption of individuals, which is needed for assessing the real risk, is difficult to estimate because it can be influenced by many factors such as age or access to drinking water resources.

16.6 Implementing MAR

16.6.1 Hydrogeology Study

The feasibility of an MAR system depends largely on local hydrogeological conditions (Dillon 2005). Understanding of natural recharge, of its evolution, and therefore of the storage capacity of the sub-surface will be a fundamental criterion for decision support in the choice of an artificial recharge site. This step of feasibility needs a closely hydrogeological analysis with the help of hydrogeologic experts that can advise about the drawbacks or benefit of the future considered MAR site.

16.6.2 Biogeochemical Processes Evaluation

In the case of artificial recharge systems that involve infiltration techniques, geo-chemical and microbiological processes might occur in the unsaturated zone that enables the purification of the recharge water. Furthermore, the unsaturated zone must allow the water to infiltrate to the aquifer, the aquifer must be able to store the infiltrated water, and then release it without excessive “dissipation”, which would cancel the storage effect.

It is, however, possible to identify the main criteria that can affect the geochemical and microbiological processes that enhance the purification of the recharge water as it moves through the unsaturated zone: (i) pH, (ii) redox potential, (iii) organic matter content, and (iv) mineralogy (Johnson et al. 1999; Rinck-Pfeiffer et al. 2000; Pettenati et al. 2012):

(i) In order to limit trace metal mobility and optimize organic contaminant degradation, it is preferable that the pH of recharge water interacting with the aquifer matrix and/or soil presents a range of values between 5 and 8. In general, adsorption processes (surface complexation and ion exchange reactions) of cations such as trace metals and degradation reactions are usually weaker at extreme pH values. Under acidic conditions ($pH < 4$), the adsorption of protons on negatively charged adsorption sites neutralises the charges of these sites, or even gives them a positive charge, which decreases the adsorption capacity of the components of the medium for cations. Under alkaline conditions ($pH > 8$), cations do not remain in the form of free ions but form aqueous complexes involving anions, usually the oxyhydroxides group, that are present in the solution.

(ii) Geo-purification processes (adsorption, dissolution/precipitation, biodegradation) are strongly influenced by the redox potential. For example, a decrease in the redox potential can cause dissolution of oxide and/or hydroxide carrier phases and therefore the release of adsorbed trace metals at their surface or in their crystal matrix. Furthermore, a decrease in the redox potential modifies the aqueous speciation of trace metals, which can increase their toxicity (for example, by transforming As(V) into As(III)).

(iii) The natural attenuation processes occurring in the soil and sub-soil, particularly in the unsaturated zone, have been shown to be quite effective with respect to trace organic removal (Ternes et al. 1998). The biodegradation process is also influenced by organic matter. Organic matter is the main source of energy for microorganisms in the unsaturated zone. In order to enhance the metal adsorption reactions and microbiological reactions including the degradation of organic pollutants or the reduction of nitrates, recharge systems should be installed on sites having significant relatively-insoluble organic matter content.

(iv) Another criterion that makes it possible to evaluate the geo-purification capacities of unsaturated zones during artificial recharge is their mineralogy. Indeed, mineralogy can strongly influence the geochemical processes that control the mobility of pollutants in the unsaturated zone. Analysing the mineralogy of the unsaturated zone makes it possible to determine the concentrations of oxyhydroxides and clay minerals, which are the solid phases that have the greatest affinity for pollutants.

16.6.3 Particular Case of SAT: Methodology of Purification Processes Evaluation

An initial, generic approach should, however, make it possible to roughly identify and quantify the potential biological activity (of the soil itself, or of the injected water) that will play a role in the evolution of the main mineral phases of interest as concerns: the physical and chemical characteristics of the soil (dissolution/precipitation), and on certain global reactions that must be defined (organic matter decomposition, redox reactions of Fe, S, Mn, etc.) depending on the nature of the injected water and the soil (Azaroual et al. 2008, 2009; Pettenati et al. 2012).

Once the MAR site has been identified, taking into account constraints such as the availability of water, hydrogeological characteristics and regulations, five steps are usually necessary:

- a preliminary evaluation of the feasibility of a recharge system on the chosen site based on existing data or modelling
- designing the recharge system
- carrying out a detailed study of the site in order to validate or supplement the results obtained in the first step
- building a pilot or experimental system at a scale that makes it possible to carry out preliminary tests
- extrapolation to an operational scale

Since the aim of SAT is to optimise the upstream treatment of residual water and the natural geo-purification of the sub-surface, a preliminary analysis of the chosen site must be carried out since the characteristics of the recharge water and of the mineralogical assemblage making up the sub-surface are site-specific. The water quality monitoring programme recommended by Ollivier et al. (2013) include the following:

- measuring physical-chemical parameters: water saturation, water pressure, temperature, conductivity, redox potential, pH of the infiltration water
- sampling and analysis of the gas in the unsaturated zone
- sampling and analysis of the water in the unsaturated zone
- sampling of the soil for mineralogical and microbiological analyses
- permeability testing of the soil and sub-soil on the recharge site

16.6.4 Cost-Benefit Study

The costs and benefits of the different management solutions (including environmental costs and benefits) must be systematically assessed in close collaboration with hydrogeological study (Shah 2014). The concept of water foot-printing needs to be deepened, establishing practical methods and certifiable systems. Innovative concepts for water resources management need to be developed, with the aim of providing science-proof solutions to societal water challenges. On the basis of French feedback (Casanova et al. 2013), the feasibility of implementing MAR strongly depends on developing new approaches for water management aiming at setting up innovative alternatives suitable for decision making. These approaches should be ideally based on: (i) the broad participation of stakeholders; (ii) multidisciplinary research; and (iii) the development of scenarios to support short to long term decision making.

16.7 Case Study of the MAR in France

Groundwater can be found in two thirds of France which has about 200 large aquifers and 6,300 small aquifers, and at least 6 billion m^3 are withdrawn every year – 59 % for drinking water supply, 19 % for agriculture (irrigation) and 22 % for industry (not including the water used by nuclear power plants) (SOeS 2012). When there is a rainfall deficit for several successive years (e.g. between 2006 and 2011), or during periods of long summer drought particularly in the southern half of France (Giuntoli et al. 2013), groundwater levels drop significantly, in particular in aquifers that are near the surface and in the large aquifers in the Paris Basin. These critical periods are usually limited in time and space.

Water resources are a crucial element in the analysis of the impacts of climate change and the suitable responses that can be proposed (Roux 1995). Indeed, climate change directly modifies both the spatial and temporal dynamics of the water cycle. The aim of the French National Plan for Adaptation to Climate Change (PNACC) is to develop concrete and operational measures to prepare France, over the next 5 years, between 2011 and 2015, for confronting and even benefitting from new climate conditions (MEDDE 2011).

The impacts of climate change on water resources are numerous (see Chap. 5) and concern both the offer and the demand, both quantitatively and qualitatively (Armandine Les Landes et al. 2014). Climate change predictions indicate that surface runoff will decrease in almost all of France's catchment basins. In particular, the decrease in runoff will be greater in areas that are already affected by structural deficits. Therefore, one of the main challenges of the future will be how to ensure the water supply that, already in some places, is not adequate and will increase due to global warming (IPCC 2014).

The Explore 2070 project aimed to determine the impacts of climate change on aquatic environments and water resources between now and 2070 in order to anticipate the main challenges to be met and rank the risks incurred (MEDDE 2013). As concerns groundwater hydrology, this project showed that there will be an almost universal lowering of the water table in France together with a 10–25 % decrease in recharge, with two zones more severely affected – the Loire catchment basin with a 25–30 % decrease in recharge over half of its surface area, and especially the Southwest of France with decreases ranging from 30–50 %.

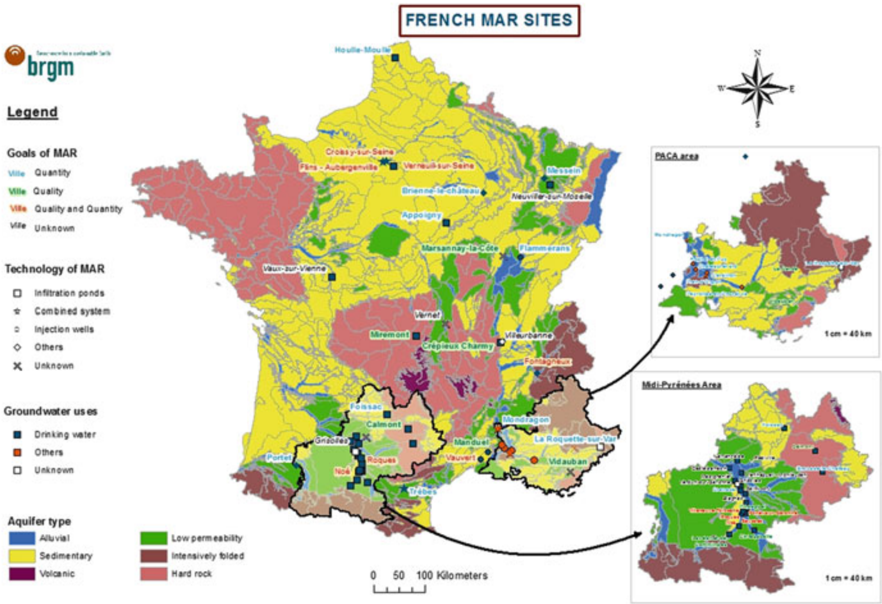

Fig. 16.1 State of MAR facilities in France

The integrated management of water resources by catchment basins must be done comprehensively, taking into account various water needs, including those of the environment. At present, it aims, within the framework of water development and management directives, to reach the objectives of the European Water Framework Directive (WFD, Directive 2000/60/EC of the European Parliament establishing a framework for Community action in the field of water policy). The anticipated impacts of climate change will affect, first of all, regions that are already encountering conflicts over water resources. It is therefore necessary to begin immediately to prevent all situations of diminishing water resources and develop strategies that promote water conservation and optimised use. MAR is one of the tools that can be used for an integrated quantitative (and/or qualitative) management of ground- and surface water resources.

A recent inventory of MAR facilities in France (Casanova et al. 2013) enabled us to identify 75 installations. The current operational status of 48 of these is known with certainty, while there is some uncertainty concerning the operational status of 8 others, and the state of 19 sites could not be determined. Two thirds of the first group are located in the Nord-Pas-de-Calais, Midi-Pyrenees and PACA regions and only about 20 of these are still active today (Fig. 16.1). Many sites have been abandoned when towns find other sources of water for their drinking water supply. In some cases, MAR was no longer needed or the quality of the recharge water no longer enabled the system to function correctly.

In most of the cases identified in France (Casanova et al. 2013), the main objective of MAR is to sustain an over-exploited groundwater aquifer. The second objective is to improve the quality of the groundwater by significantly decreasing the concentrations of some chemicals by dilution (e.g. nitrate, pesticides), thereby enabling the use of simpler and cheaper water treatment methods to reach drinking water standards. More precisely, in France, the objective most often sought in MAR projects is quantitative. More than half of the sites inventoried by Casanova et al. (2013) had a quantitative objective, a quarter of them had no clearly defined objective, while the others aimed at improving water quality or had an objective that was both qualitative and quantitative.

MAR is also undertaken to protect the environment by limiting the level of pollution in sensitive receptor media. In 2009, 41.4 % of France's surface water bodies was assessed as having a good ecological status and 43.1 % a good chemical status. In addition, 58.9 % of its groundwater bodies possessed a good chemical status and 89.4 % a good quantitative status (MEDDE 2012a). The quality of France's groundwater is better than that of its surface water (60 % of the groundwater bodies in France and 80 % in Europe having "good" chemical status). For this reason, groundwater is often used as a source of drinking water. However, the number of French groundwater bodies that have been disqualified for drinking water supply due to nitrates and pesticides is rather large (higher than the European average, the cause of the poor status being divided equally between pesticides and nitrates) (MEDDE 2012b). In France, MAR is therefore often used to dilute pollution in groundwater bodies that are tapped for drinking water such as the MAR installation of Lavelanet-de-Commingues (Haute Garonne, France). This MAR is assigned to decrease the nitrate concentration (>50 mg l^{-1}) of groundwater. Recharge water is abstracted from the upstream Tuchan canal and transport to the water catchment in decantation ponds previous to infiltration ponds. This system permits to maintain a nitrate concentration in the groundwater around 30 mg l^{-1} (Wuilleumier and Seguin 2003).

Conversely, in France, MAR can also be used to limit the pollution of surface water by infiltrating some of the polluted water and monitoring the geo-purification processes. MAR is used to manage stormwater in many French cities where surface runoff is collected in infiltration basins. However, stormwater is one of the main sources of pollutants (heavy metals, hydrocarbons and other organic compounds) produced by cities. In consequence, traditional urban drainage systems now cause many technical and environmental problems, notably the pollution of the surface receptor media (Chocat et al. 2007).

16.8 Conclusions

Recurrent water resources crises call for a better understanding of hydrological processes and improved technical and socioeconomic groundwater management. In many areas of Europe, including France, growing freshwater scarcity currently emphasizes the need to close the water cycle gap by reconciling water supply with demand both in quantity and quality terms. The demand for closed water systems is obvious in semiarid areas, where research institutes are currently developing new concepts and technologies. MAR is one of the strategies that can be used for quantitative and qualitative water management and adaptation to climate change in the field of water resources. The various methods used at the sites currently in activity in France and elsewhere in the world use technologies that, for the most part, have been relatively well perfected over the last 20 years.

Water resources observation and modelling are required to better understand hydrological processes and to analyse and forecast the effect of management options. This technological and environmental research must be systematically combined with a socio-economic approach investigating the questions of participation, behaviour and commitment of stakeholders. The choice of a method for artificial recharge depends on numerous factors such as the objective (quantitative and/or qualitative), the local hydrogeological context, the type and volume of recharge water available, and the chemical and microbiological characteristics of this water. Laws regulate the construction and operation of recharge systems. One criterion common to all identified artificial recharge French sites is that they have all been built using a multi-step procedure. Independently of the social, economic and environmental impact that must be taken into account, this chapter highlights the challenge that must be overcome upstream of any regulatory modifications that aim to facilitate the use of these technologies.

Because of the specific local characteristics of each MAR site (Fig. 16.2), there is no universal solution that can be recommended and any change in laws must take this into account. It seems, however, possible to break down artificial recharge installations into two groups based on the quality of recharge water. Water whose quality is similar to drinking water standards is better-suited to direct or indirect injection into the aquifer, whereas for water whose quality is degraded, preference should be given to infiltration methods that enhance additional natural treatment in the subsurface. In both cases, post-treatment, the intensity of which depends on the foreseen use of the pumped water, is necessary before distribution.

Therefore, the initial objective is often to sustain an over-exploited aquifer with other induced benefits such as improved groundwater quality with a significant decrease in the concentrations of some reactive chemicals (i.e. iron, manganese, ammonium, nitrate, organic pollutants, etc.). This enables the use of simpler and cheaper water treatment methods to reach drinking water standards. At the same time, the contamination of the infiltrated water can be reduced naturally if the procedure used to site the installation includes the identification of reactive zones and/or buffer zones and zones that are favourable to the development of microorganisms. Indeed, clay minerals, iron and manganese hydroxides, and microorganisms present in the different zones have great capacities for decontaminating (i.e. biodegradation of organic compounds, etc.) and fixing metallic pollutants and metalloids. Choosing a MAR site therefore requires that the quality of the recharge water is compatible with the soil's reactive processes, especially in the unsaturated zone. In this context, the final treatment of the water might be optimised and hence become less costly.

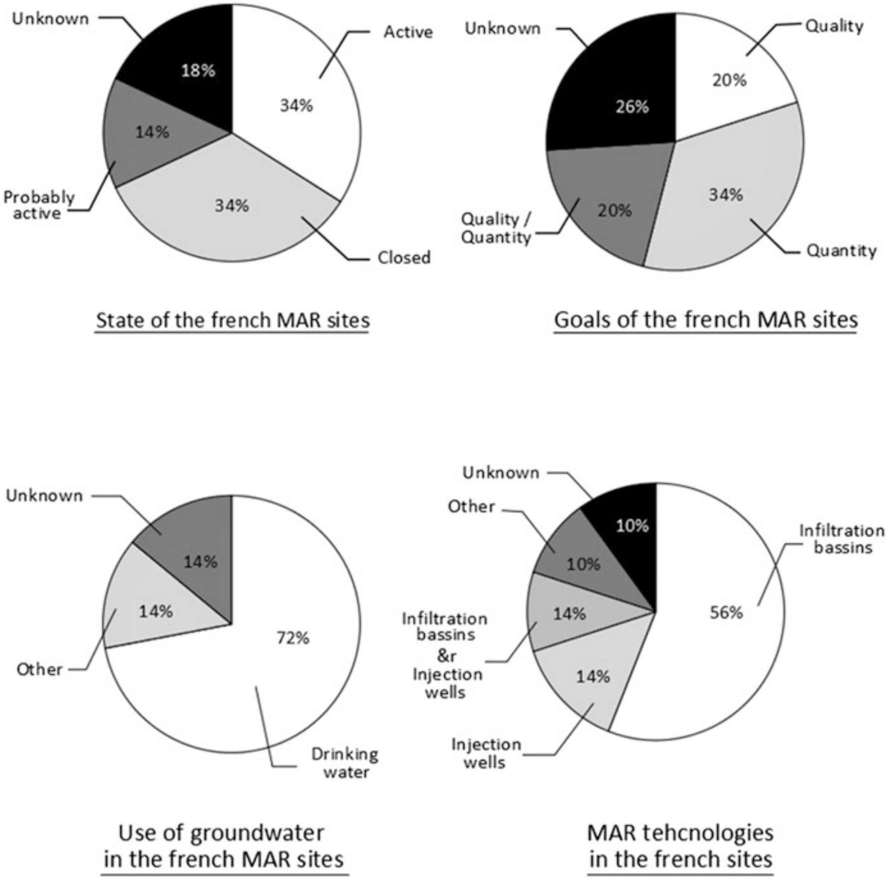

Fig. 16.2 Characteristics of the French MAR sites

In order to meet WFD challenges, links between pressures and water resources have to be established through research activities aiming at elucidating specific connections between water resources, pressures and uses. The combination of observations and hydrological modelling (water bodies, overland flow, unsaturated zone, groundwater and land cover) might be targeted to ensure proper conceptualization of the involved processes. In Europe MAR implementations are being widely reapplied and developed using current technologies. However, French examples of quantified assessments of their effectiveness are limited. Improved understanding of how recharge structures actually function and the impact they have on water availability, water quality, sustainability as well as on the local and downstream environment, need to be gained and disseminated to promote cost-effective implementation.

It is generally assumed that MAR systems will be used throughout the world, including in France, due to the fact that MAR is a pragmatic and potentially eco-responsible response to climate change, and our need to adapt to it in a systemic approach to environmental management. Moreover, it is economically attractive for water resource management. It is however difficult to quantify the exact cost of the construction, use and profitability of these systems. Regardless of which technical solution is chosen for recharge, the costs of pre- and/or post-treatment depend strongly on the quality of the injected water. Systems that give preference to slow infiltration and an optimisation of the geo-purification capacities of the sub-surface, therefore, make it possible to minimise the costs inherent in recharge water treatment and enhance the profitability of the project.

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Managed Aquifer Recharge in Integrated Water Resource Management

17

Peter Dillon and Muhammad Arshad

Abstract

Managed aquifer recharge (MAR) is one tool in integrated water resources management which can restore over-allocated or brackish aquifers, protect groundwater-dependent ecosystems, enhance urban and rural water supplies, reduce evaporation losses and improve water supply security. This chapter describes the ways in which MAR is used around the world and presents two Australian case studies, with a focus on economics. Aquifer storage and recovery of urban stormwater via a confined limestone aquifer is shown to provide a viable alternative to use of existing mains water or desalinated seawater for public open space irrigation. The second case study is a desk-top evaluation of the potential for recharge of harvested floodwater via infiltration basins for irrigation of cotton and faba bean crops. Based on assumptions about scale of operations, component and maintenance costs, and evaporation losses, the net benefits of infiltration basins for a range of infiltration rates were compared with those of surface water storage and of aquifer storage and recovery wells. Infiltration basins with moderate to high rates of infiltration (>0.15 m/d) had the highest net benefits and warrant testing in a pilot program. Water treatment costs make ASR with flood waters unattractive for crop irrigation, in comparison with both basin infiltration and surface storage. Selection of the most economic method of storage depends on availability of an aquifer, soil and subsurface hydraulic characteristics, available quantity and quality of surface water, land value and end use of the water. MAR is shown to offer a range of options that

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warrant investigation in comparison with conventional supply alternatives to enable the most effective water resources management to be implemented.

17.1 Introduction

This three-part paper describes the role of managed aquifer recharge (MAR) in integrated water management, and then provides two case studies. The first is the storage of urban stormwater for later reuse to irrigate public open space in the City of Salisbury in South Australia. This has been practised in a confined brackish limestone aquifer for 20 years and the number of aquifer storage and recovery wells continues to grow. The second case study is a desktop evaluation of the potential for storing flood water in a rural catchment to allow irrigated crop production to be expanded or to achieve environmental goals by replenishing a depleted aquifer (Rawluk et al. 2013; Arshad et al. 2012). That desktop study suggested that managed aquifer recharge via infiltration basins may be viable.

17.2 Managed Aquifer Recharge to Date

Managed aquifer recharge is defined as the purposeful recharge of water to aquifers for subsequent recovery or for environmental benefit (Dillon et al. 2009a). MAR may be used to replenish depleted aquifers, in association with demand management strategies to bring aquifers back into hydrologic equilibrium while minimising adverse impacts on livelihoods of irrigation communities. A series of examples from India and Australia are shown in Dillon et al. (2009b) that illustrate coupling MAR with demand management to achieve groundwater supplies with aquifer storage hydrologic equilibrium. Managed aquifer recharge augments groundwater with available surface water and acts alongside conjunctive use of surface waters and groundwater to sustain water supplies and achieve groundwater and surface water management objectives such as protection of ecosystems (Fig. 17.1).

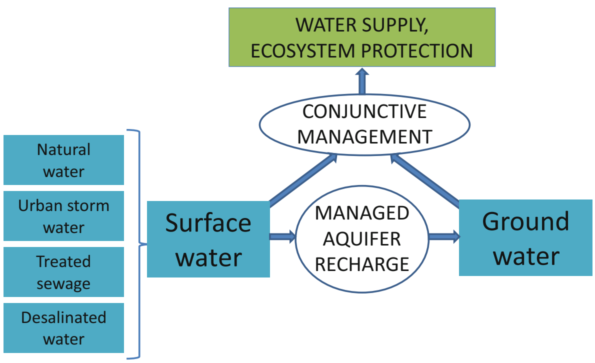

Fig. 17.1 Roles of managed aquifer recharge and conjunctive use in integrated water resources management

There are countless examples around the world that demonstrate the value of managed aquifer recharge. India leads the world in recharge enhancement with about 3 km^3/year, almost exclusively to unconfined aquifers through infiltration structures to help sustain groundwater supplies predominantly for agriculture and increasingly in urban areas. This volume does not keep up with groundwater storage depletion in northern India, but does help to prolong the resource and allow a window of opportunity for adaptive management. Water quality is rarely intentionally managed so it can be claimed that this recharge is not yet managed aquifer recharge. The same can be said for many parts of the world where untreated sewage and industrial effluent, stormwater or blends are allowed to infiltrate and contaminate aquifers and diminish the useable resource. If appropriately treated, this water would have supply benefits as well as environmental and health improvements.

Roof top rainwater and urban stormwater have been recharged in Australia, Germany, India, Jordan, USA and in many locations with permeable soils or karstic aquifers. There is now a progression underway from uncontrolled disposal via sumps, basins, wells and karst features to managed aquifer recharge through implementing measures to improve and protect water quality. In coastal locations in California, China, and Bangladesh replenishment of aquifers using injection wells has protected urban and irrigation supplies from salinization and in some places has been claimed to assist in mitigating against land subsidence. Treated sewage effluents have been used to augment and secure groundwater supplies in Australia, Belgium, Germany, Israel, Italy, Mexico, Namibia, South Africa, Spain, USA and elsewhere. Desalinated water is also used in UAE and USA for recharge primarily to build secure reserves of mains water. In a few locations groundwater from one aquifer is stored in another to secure supplies.

Riverbank filtration is another widespread technique to improve water quality and security of drinking water supplies. Being a low energy method for water treatment its popularity will grow as the treatment effectiveness of alluvium becomes better understood. Recharge has also been practiced for protection of groundwater dependent ecosystems (Berry and Armstrong 1997; Dillon et al. 2009c). There are many technical papers on managing aquifer recharge available from the IAH-MAR web site www.iah.org/recharge and some of these (in English and Spanish) are stored on our companion Spanish web site accessed from the same URL.

Figure 17.2 demonstrates how managed aquifer recharge can act alongside demand management and conjunctive use to bring an over-exploited aquifer back into hydrologic equilibrium. A corollary of this is that in areas where the climate is drying, causing natural recharge rate to decline and irrigation demand to increase, managed aquifer recharge may provide an adaptive strategy to help re-establish hydrologic equilibrium.

It is logical that at any location the most economic option available would be adopted first, and then the next most economic, and so on until the volume by which demand is decreased, or the volume of managed aquifer recharge or supply substitution is increased so that hydrologic equilibrium is achieved. Invariably, some strategies for increasing water use efficiency will be among the most economic options.

Figure 17.3 represents the actual sequence of options for restoring the aquifers of the Swan Coastal Plain and continuing to supply Perth's growing need for water.

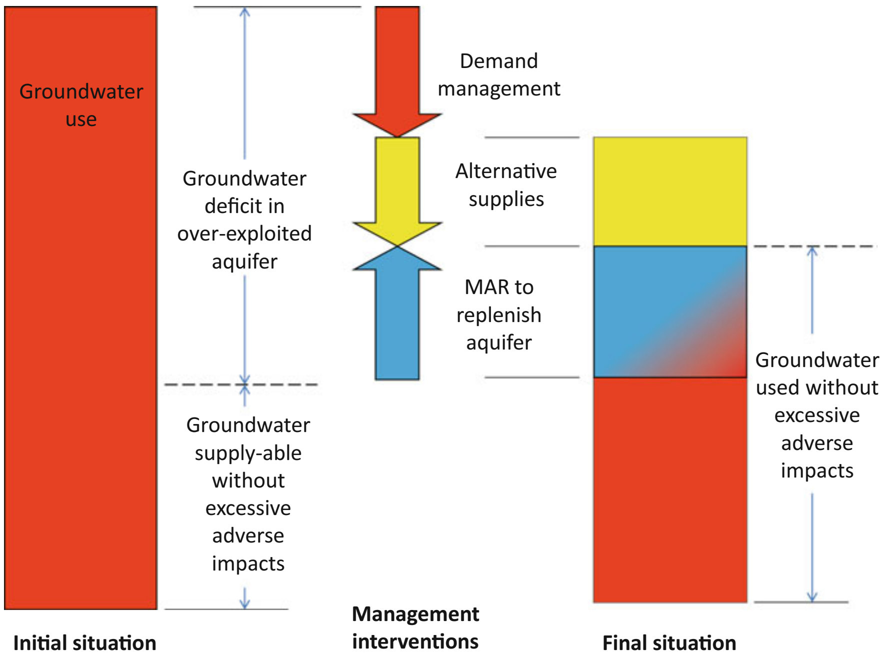

Fig. 17.2 An aquifer can be brought into hydrologic equilibrium by either reducing extraction, or augmenting supplies, either through groundwater replenishment or providing alternative supplies (conjunctive use) (From Dillon et al. 2012)

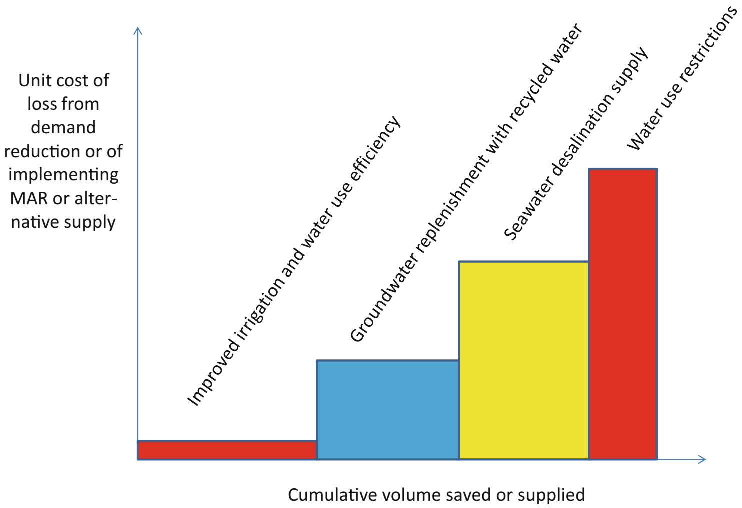

Fig. 17.3 A logical sequence of demand reduction (*red*), MAR (*blue*) and conjunctive use (*yellow*) to reign in groundwater depletion and sustain supplies (in this case for Perth, Western Australia). Options, their relative costs and volumes are location-specific. Improved irrigation efficiency is frequently the least costly option and should be implemented first (Adapted from Dillon et al. 2012)

The Water Corporation in the state of Western Australia imposed a series of water conservation measures, such as mandatory restrictions on the hours during which parks and gardens may be irrigated to avoid high rates of evaporation, and reducing the frequency of irrigation to once or twice a week. It also encouraged water efficient household appliances such as washing machines, showers, and toilets. An extensive investigation and demonstration trial of the use of recycled water for groundwater replenishment was undertaken, and the Western Australian Minister for Water announced in 2013 that this would be the next water supply for Perth, based on safety, economic efficiency and public acceptance. The value of the research was that it showed the costs of groundwater replenishment with recycled water were approximately half those of harnessing seawater desalination, the alternative (conjunctive supply) previously regarded as the cheapest acceptable source of supply. Prior to that, onerous water restrictions were the only option, and these were regarded as politically unsustainable, and caused failures in garden supplies industries.

This framework of integrated water management is used in this chapter to explore two case studies of the potential for managed aquifer recharge, one harnessing urban stormwater in a city for public open space irrigation and industrial use. The other is a desktop study for a rural area that assesses the opportunities to harvest from large floods in order to sustain agricultural irrigation. These studies focus on the economic aspects of MAR in relation to alternatives.

17.3 Potential for Managed Aquifer Recharge from Urban Stormwater in a Suburban Area of SA, Australia

The driver for this case study is not the need to reduce groundwater demand. In fact the aquifer originally contained brackish groundwater and demand was negligible prior to aquifer storage and recovery (ASR) with stormwater. The purpose of recharge was to store fresh urban stormwater runoff during wet winters and recover it for irrigation in dry summers in an area with a Mediterranean climate. The costs of MAR water supplies for local councils were cheaper than the costs of purchasing mains water from the state government-owned water utility. This supply met with the approval of the State at that time as it reduced demand on drinking water supplies and hence had a positive effect on the security of those supplies in a system that had little storage capacity and was drought-prone.

The costs of producing these supplies were calculated in AUD 2008 as shown in Table 17.1, based on data from consultants and owners of eight ASR systems with capacities between 75 and 2,000 ML/year. Costs exclude value of land occupied by wetlands used for water harvesting. In all cases the proponent of the project already owned this land. In most cases a wetland was required as a detention basin to prevent increased peak flow rates during storms as a result of new urban developments. Hence the land for the wetland was considered as contributing to the flood mitigation benefit, and the remaining costs, including wetland

Table 17.1 Mean levelised costs (in AUD 2008) for components of urban stormwater ASR projects for irrigation supplies in the size range 75–2000 ML/year (Adapted from Dillon et al. 2009a)

Project component	Number of sites with costs	Component cost as % of total cost	Mean levelised cost (A$/kL)
Investigations	7	11	0.12
Capital costs of water harvesting	5	25	0.28
Capital costs of treatment, ASR, distribution	5	39	0.44
Total capital costs	8	64	0.72
Total initial costs (minus land)	7	74	0.84
Operation, maintenance and management	8	26	0.28
Total levelised cost (minus land)	8	100	1.12

construction, were attributed to producing a water supply via ASR. Levelised cost, expressed in $/KL, was calculated as annualised cost to amortise capital cost components over their expected working life added to the annual operating and maintenance expenses and divided by the annual volume of supply. In this case for eight stormwater ASR projects in South Australia, the adopted discount rate was 7 % and the working life of ASR wells was assumed to be 15 years, for wetland systems 25 years, and for distribution systems 50 years. It was also assumed that only 80 % of injected water could be recovered at the salinity required for its intended use.

The mean levelised cost for ASR (A$1.12) compared favourably with independently provided figures by consultants for two seawater desalination options ranging from A$2.45 to 3.76/KL levelised cost. The ASR energy intensity of 0.10 KWh/KL compared favourably with seawater desalination and distribution of 4.2 to 5.3 KWh/KL (Dillon et al. 2009a). That is, the mean levelised cost from the sample of stormwater ASR projects was found to be between 30 % and 46 % of that of seawater desalination, and greenhouse gas emissions were less than 3 % of seawater desalination.

Levelised costs for ASR reduced as recharge rate increased. The eight projects costed had injection rates from ~10 to ~30 L/s per well. Hence sites with higher well yields and transmissivities are preferred. For low permeability formations the levelised costs of recharge are elevated due to the capital and operating costs being amortised over smaller volumes of water and because additional water treatment may be required in order to avoid clogging of the well. An example in south-east Melbourne is reported in Dillon et al. (2010) where levelised costs of ASR exceed A$8/KL in a formation with a transmissivity of ~1 m^2/day, sustaining an injection rate of 0.4 L/s and requiring ultrafiltration and granular activated carbon filtration as pre-treatments to avoid clogging for recharge of 4ML/year.

A more recent study of stormwater recharge on the Northern Adelaide Plains (Dandy et al. 2013) revealed levelised costs of A$1.57/KL (in 2012–2013) including land value of the harvesting facility and capital and operating costs of the distribution system for public open space irrigation. The same study found that recovery for potable use of treated stormwater had a levelised cost of between A$1.47 and A$2.51/KL depending on whether the water was pumped to an existing dam and treatment plant or was treated locally in a decentralised treatment plant. These costs include the costs of treatment and implementation of a risk-based management plan appropriate to the end use. Equivalent financial results for water recycling from treated sewage effluent via aquifers to various end uses will be available in 2015 from the Australian Water Recycling Centre of Excellence.

ASR sites with higher ambient groundwater salinity generally allow a smaller proportion of injected water to be recovered at a salinity that is acceptable for its intended use. This is exacerbated where native groundwater has sufficiently high salinity that density-affected flow occurs (Ward et al. 2009) and a freshwater injection lens forms at the top of the aquifer. This is difficult to recover without also entraining some of the saline water underneath. Recovery efficiency therefore also influences the levelised cost of ASR operations and needs to be taken into account wherever the native groundwater is not fit for the intended use of recovered water.

17.4 Potential of Managed Aquifer Recharge from Large Floods Events in a Rural Irrigation Area of NSW, Australia

Groundwater in the Namoi River Catchment in the Australian state of New South Wales supports an irrigation industry worth in excess of AU$ 380 million per annum (Namoi CMA 2013). According to The Australian Cotton Grower (2012), in the wetter year of 1998/1999 about 60,000 ha of cotton were grown in the Lower Namoi, whereas in the drought year of 2003/2004 only 26,300 ha were planted due to limited surface water supplies.

In response to groundwater overdraft, State governments in Australia have reduced current groundwater irrigation entitlements in stressed aquifer systems (Smithson 2009). For the Lower Namoi Valley, a highly developed cotton irrigation district in NSW, this gradual cutback of 10 % each year, translates to a reduction of 21 gigalitres (GL)/year in groundwater entitlements for irrigation by 2015 and beyond. Reduced water availability under droughts and reduction in water allocation have significant financial impact on the farming communities.

A typical Namoi valley farm holds enough water in storage (600–900 ML) for 1 year of irrigation (Powell and Scott 2011). All irrigation water is stored and routed from surface storages before application to the field, resulting in substantial evaporation losses. On average, evaporation losses from surface water storages range between 1.2 and 1.8 m/year (Wigginton 2011), this represents a loss of approximately 35 % to 50 % of the total on-farm storage capacity.

Aquifer storage via Managed Aquifer Recharge (MAR) was investigated as a way of minimising evaporative losses and increasing farm profitability. MAR can serve the purpose of increasing groundwater storage in wet periods in order to support irrigation and environmental use of water during dry periods. The case study highlights the availability of water from high flood events that may be used for aquifer recharge and examines the financial costs and benefits of storing floodwater underground (via infiltration basins or injection wells) compared with the current method using surface storages.

Assessing the feasibility of MAR requires the integration of many types of data and information from many disciplines to assessing hydrologic, hydrogeologic, social, institutional factors and environmental risks. Carrying out a comprehensive feasibility assessment is essential; the first step in establishing an MAR scheme requires assessing the feasibility of technical and economic factors, to provide a basis for other investigations to proceed. An overview of the basic requirements and feasibility guidelines for MAR is available in Dillon et al. (2009a)

Before conducting costly technical feasibility studies through geophysical and hydrogeological investigations, a first step is to explore the potential of MAR through a desktop case study to address two questions that are of major concern to the irrigation farmers of the Lower Namoi;

I. Is a reliable source of water for aquifer storage available? and
II. Is underground storage financially better than surface storage?

Identifying water for the purpose of MAR will be challenging, particularly in the Murray-Darling Basin where irrigators must operate within existing entitlements to water and where flood waters are typically considered as environmental water. Under these arrangements only existing entitlements for consumptive use can realistically be considered as a source of water for MAR in rural catchments. Within existing entitlements for consumptive uses, Rawluk et al. (2013) discussed the potential sources of water for MAR in the Murray Darling Basin.

The potential sources for MAR water in the Lower Namoi may include;

i. Water diverted from rivers under existing entitlements to take water during high floods or periods of high streamflow, known as supplementary water.
ii. Locally captured farm run-off.
iii. Water used in coal seam gas mining could be treated (desalinated) and reused for MAR.

In many areas of Australia, including the Lower Namoi, supplementary water and local run-off is captured and stored in farm dams for stock supply and irrigation. Currently, farm dams across the Murray-Darling Basin have a combined capacity of 2,000 GL (CSIRO 2007). Craig et al. (2005) estimated that up to 40 % (800 GL) of this storage volume can be lost each year to evaporation. Most situations in which there are opportunities for such water capture are on floodplains in the lower parts of major catchments, including the Lower Namoi. In these situations the alluvial sediments offer storage opportunities either through surface recharge or deep injection into alluvial aquifers, depending on local hydrogeology. MAR can provide a low or no evaporation option for storage of water under these circumstances (Ross and Arshad 2013); however some water may not be recoverable, termed as recovery losses, if native groundwater is not of a suitable quality for irrigation or if there is leakage from the aquifer to other aquifers or to surface water bodies.

The Namoi River follows an irregular flow pattern with moderate to large variability in inter-annual and inter-decadal flows. Figure 17.4 highlights floods from the river flow data (1970–2008) at the Mollee gauging station where recorded mean daily flows in months during flood events in 1964, 1971, 1974, 1976, 1984, 1998 and 2000 were between 100 and 200 GL/day. This is a huge volume of water when compared to the long term median flow of 0.53 GL/day, indicated by the horizontal bar in Fig. 17.4. Such peaks could be more frequent in future under climate change (Barron et al. 2011; Chiew et al. 2011).

From the flood frequency and magnitude data, it appears that a significant volume of water could be made available for MAR provided that environmental flows and ecological requirements are met. The Water Sharing Plan (The Plan) for the Upper Namoi and Lower Namoi Regulated River Water Sources (NSW DIPNR 2003) states the extraction rules for supplementary water entitlements held by irrigators. Under The Plan, flood water that is not already allocated is assumed to be environmental water, except that holders of the supplementary access licences can extract water during the announced supplementary periods. Such access periods are typically during floods and periods of high streamflow, when dams spill and flows are in excess of licensed obligations and environmental needs (Burrell et al. 2011).

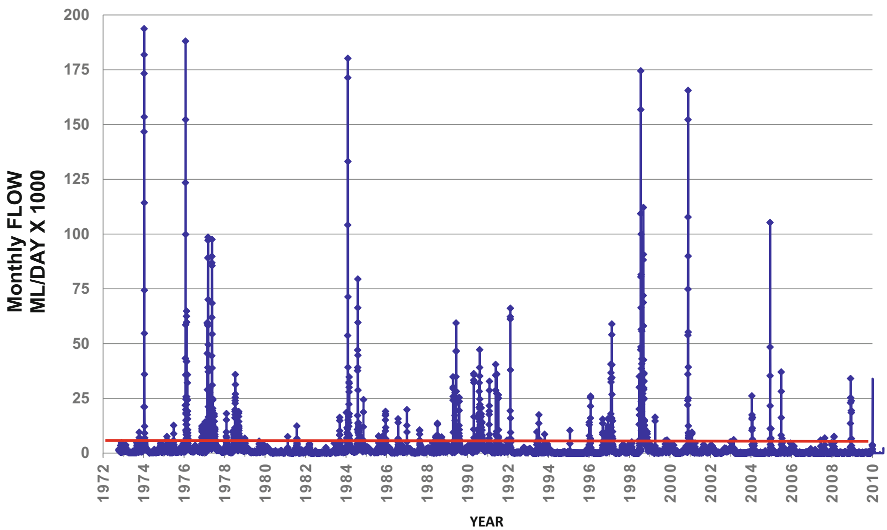

Fig. 17.4 Flood peaks in the Namoi River at Mollee (1970–2008) (Source: NSW, Office of Water 2008)

Under The Plan the volume of water that may be taken from a supplementary access event depends on the time of year. The Plan provides all the basic rules for capture of water during each supplementary event. However, the rules defining the threshold for the announcement of a supplementary access event are complex and depend on many factors. The rules in The Plan detail the various start, stop and flow triggers for different locations and the different scenarios that apply depending on the volumes of water allocations in the regulated river. In addition, the available volume of water for extraction varies for different times of the year, that is up to 10 % of the event volume between 1 July and 31 October and up to 50 % during other times. A water user is only able to extract supplementary water when, amongst other things, their supplementary water account balance is in credit.

After meeting all other requirements, supplementary access is only available when the uncontrolled flows are surplus to other needs and is only permitted in accordance with announcements made by the Minister's Office of Water. Arshad et al. (2012) made a quantitative assessment of the volume of water from high flow events. This was achieved by analysing daily streamflow data (NSW, Office of Water 2008), from 1972 to 2012 at the Mollee gauging station.

In the absence of any published threshold volume that could be used to establish the start of a supplementary event, Arshad et al. (2012) adopted a threshold of 37.8 GL/day. This threshold was based on the peak flow of the Namoi River on 1 August 2011 at Gunnedah when the river level was more than one metre higher than the river bank (Burrell et al. 2011). With this level of inundation in the floodplain it is assumed that all the basic environmental and ecological requirements are met locally and downstream.

Figure 17.5 shows the share of irrigation and environmental water for each of the supplementary water events in the Lower Namoi from 1972 to 2012. As is indicated

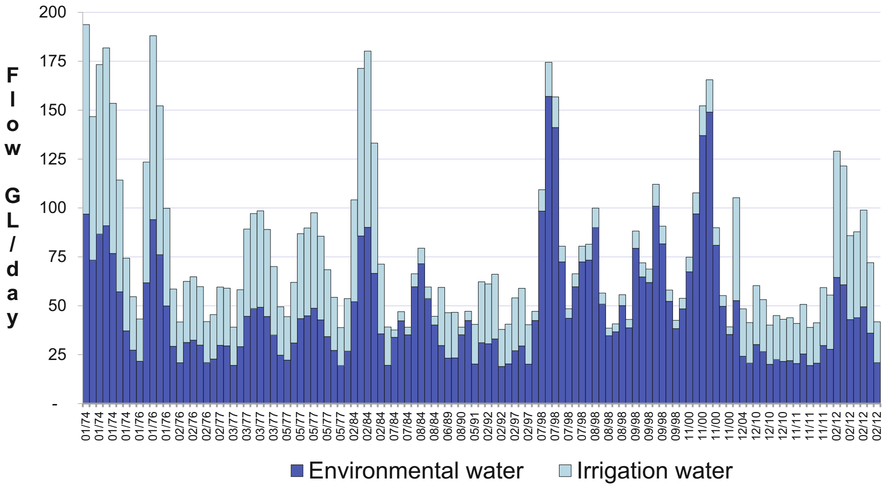

Fig. 17.5 Shares of irrigation and environmental water (1972–2012). Supplementary access set at a threshold of 37.8 GL/day, Namoi at Mollee (Data Source: NSW, Office of Water 2012)

in Fig. 17.5, in the 40 years between 1972 and 2012, there were 120 high flow events exceeding the threshold of 37.8 GL/day. These flows provided an average 85 GL of water per year for irrigation that may be available for aquifer storage which is a significant amount of water.

There is sufficient space in the main aquifers in the Lower Namoi to store this volume of supplementary irrigation water. Historical groundwater extraction, supporting the irrigation industry since the 1960s, has been in excess of groundwater recharge. This has generated a huge storage space within the alluvial aquifer. The captured supplementary water could be placed in either the shallow unconfined aquifer, or the semi-confined aquifers from which the irrigation bores extract groundwater.

Periods of high streamflow and floods offer a significant opportunity for diverting river water, and storing it in the aquifers of the Lower Namoi. However, the amount of flood water is highly variable from year-to-year, ranging from 11 GL in 1990 to 691 GL in 1977. This large variability in the volume of flood water will require temporary storages to capture, stabilise and/or treat the water before recharging it into the aquifers. Before establishing institutional mechanisms to implement MAR in Lower Namoi, a financial comparison of the costs and benefits of surface storage and underground storage using MAR would be needed.

The cost of MAR depends on number of factors such as local hydrogeology; e.g. infiltration and injection rates, cost of physical infrastructure and its maintenance, cost of acquiring source water, level of required water treatment, cost of land and cost of pumping to recover stored water. Arshad et al. (2013) carried out a cost-benefit analysis of surface and aquifer storage of 600 ML/year in Lower Namoi for a typical cotton irrigation farm. The study estimated all the irrigation related costs and benefits and compared net irrigation benefits under three different water storage scenarios: surface storage in farm dams, aquifer storage using basin infiltration, and aquifer storage using aquifer storage and recovery (ASR) wells. In a typical Lower Namoi farm all the surface water allocations, including flood water, is stored in farm dams before application to the fields.

Surface storages have significant evaporation losses reported as high as 35–45 % from surface farm dams annually (Craig 2006; Craig et al. 2005). MAR can be an option to minimize evaporation losses by storing water in aquifers and recovering that water when needed. This would allow additional land to irrigate with saved water and possibly additional farm benefits. Increased costs are however incurred on establishing MAR infrastructure and its ongoing operation and maintenance. The annual irrigation water allocation from all sources for an average cotton farm in Lower Namoi is approximately 1,350 ML. However, in this analysis we only consider and report costs and benefits of 200 ML of flood water, which is only 25 % of flood water allocation and is based on recent statutory allocations of flood water (800 ML/year) in the study area.

One limitation of the study of (Arshad et al. 2013) was that it assumed average basin infiltration and ASR well injection rates that could be possible in areas with favourable hydrogeological conditions and may be uncertain at other places due to hydrogeological heterogeneity. The following section extends the analysis by considering a range of infiltration rates. The analysis also considers the comparative cost advantage of using an existing borehole for an ASR facility.

17.4.1 More Detailed Costings for the Case Study

Cost estimates of aquifer recharge are scarce and can vary considerably with location. Itemized costs for this study which are identified in subsequent paragraphs were estimated by combining current market rates of earthworks, services and materials for water infrastructure projects in Australia and were adjusted to the local situation and market rates in the Lower Namoi. Cost estimates were also compared with published data and technical reports including Khan et al. (2008), Dillon et al. (2009a) and Pyne (2010).

Capital costs of basin infiltration were estimated by assuming a range of infiltration rates (0.1–0.3 m/day) and calculating the required land area to achieve 2 ML of recharge per day. The target flood water harvested volume of 200 ML would generally appear in four or more episodes in a flood year. The flood water is collected and temporary held in farm dams before recharge. An infiltration pond with surface area of 1 ha and infiltration rate 0.2 m per day would recharge 50 ML of floodwater in a cycle of 25 days. For an infiltration rate of 0.1 m/day a pond with surface area of 2 ha would be required to recharge 50 ML in the same period.

The cost of underground storage primarily depends on the hydrogeological features of the target aquifer and the choice of method considered suitable to accomplish recharge. Apart from quality of source water, infiltration and injection rates can highly influence the cost of any aquifer recharge and storage facility. Bouwer (1999) provides typical infiltration rates for surface infiltration systems in the range from 0.3 to 3 m/day with relatively clean and low turbidity river water. For systems that are operated year-round, long-term infiltration rates vary from 30 to 500 m/year, depending on soil type, water quality and climate.

ASR can potentially achieve injection rates between 0.5 and 8 ML/day per borehole. In a modelling study Khan et al. (2008) assumed an injection rate of 8 ML per day per borehole for an ASR facility in the alluvial aquifers of the Murrumbidgee catchment. In the absence of accurate well injection rates based on field monitoring, Pyne (2005) observed that the injection rates of ASR increase with increasing aquifer transmissivities. For the Lower Namoi Williams (1989) reported that the main aquifers which are tapped for irrigation extraction are associated with the Gunnedah and Cubbaroo Formations with transmissivities in the range of 1,000–2,000 m^2/day. The yields from bores tapping these aquifers vary up to 250 L/s in the Gunnedah Formation at depths of 60–90 m and in the deep Cubbaroo Formation at depths of 80–120 m. The shallow Narrabri Formation has transmissivities less than 250 m^2/day. For this study an assumed injection rate of 25 L/s (2.2 ML/day) is considered likely for an ASR well.

The analysis assumed 40 % evaporative losses from surface storage and 5 % from basin infiltration and ASR. In the base case the only cost considered is the cost of harvesting 200 ML of flood water and the cost of annual maintenance of the farm dam. The capital cost of basin infiltration includes the cost of earth works and laying of pipes. Ongoing costs include operation and maintenance of water harvesting and recovery and the cost of basin de-silting. An existing bore is assumed to be available for recovery after basin infiltration or for injection and recovery in ASR. The capital cost of an ASR facility on an existing farm primarily includes setting up a coagulation and filtration pre-treatment facility, with capital cost assumed as A$ 250/ML. Ongoing operation and maintenance costs for ASR include well flushing and cleaning, flood water harvesting, water treatment and recovery. The analysis assumed a 30 year life span for surface storage and basin infiltration and 20 years for ASR and 7 % uniform discount rate for all options. All capital costs estimates are exclusive of land value.

With the additional 70 ML of water saved from evaporation through MAR, farmers in the Namoi have a choice to irrigate additional land with cotton, faba bean or some combination of the two crops that yields the highest returns. Value brought by the flood water under each option is estimated from the useable volume of flood water, after evaporative and recovery losses, times the gross margin per megalitre of mixed cropping of cotton and faba bean on equal land area. On average, for a typical lower Namoi irrigation farm average gross margins for cotton and faba bean are estimated as $310 and $435 respectively, averaging $ 342.3/ML of irrigation water. Details of farm benefits are available in Powell and Scott (2011) and their estimation is in Arshad et al. (2013). Table 17.2 summarises the costs and value addition of 200 ML of flood water with each water storage option in A$/ML.

The cost and value addition of basin infiltration depends highly on the infiltration rates; as the infiltration rates increase the capital costs decrease and value of saved water increases. Basin infiltration at an infiltration rate of 0.10 m/day proves to be uneconomical with 15 % less benefits than surface storage. With infiltration rates of 0.15 m/day basin infiltration is marginally profitable, while with infiltration rates of 0.2 m/ day and above basin infiltration becomes economically viable. The break-even point, where the added value of basin infiltration exceeds the additional costs occurs at an infiltration rate of 0.14 m/day.

In the Lower Namoi, areas with floodwater infiltration rates of 0.2 m/day and above can potentially benefit from aquifer storage of floodwater using basin infiltration. Basin infiltration systems could be piloted in areas where river-aquifer connectivity exists, particularly in zones where the river system is losing to the aquifer. Basin infiltration systems could be feasible to recharge unconfined shallow aquifers. A high cost of treatment of relatively turbid floodwaters was conservatively assumed for ASR, although testing is warranted to determine the level of treatment required for sustainable operation. Under the current assumptions even using existing wells, ASR appears to uneconomical due to the high cost of water treatment.

In the Lower Namoi, the opportunity for aquifer storage can be advantageous for two reasons: (a) under existing rules, large quantities of floodwater are available to harvest in wet periods and can be stored underground; (b) the existing on-farm storage dams avoid the need for building temporary storage of floodwater before

Table 17.2 Levelised costs and farm benefits (A$/ML) of Surface storage and MAR

	Unit	Surface storage	Basin infiltration rate					ASR with existing well
Infiltration rate (m/day)			0.1	0.15	0.2	0.25	0.3	2.2ML/day
injection rate (ML/day)								
Volume taken under supplementary entitlement during flood event	ML	200	200	200	200	200	200	200
Useable volume (after losses)	ML	120	190	190	190	190	190	190
(a) Annualised cost of capital items	$/ML	0	63.8	42.6	32.2	25.7	21.3	26
(b) Annual cost of operation, maintenance and management	$/ML	26	108.5	97.9	90.5	80.7	76.1	221.7
(c) Total Annual Cost (a + b)	$/ML	26.0	172.3	140.5	122.7	106.4	97.4	247.7
Gross value of crop $/ML	$/ML	342.3	342.3	342.3	342.3	342.3	342.3	342.3
(d) Value of crop that can be grown with the useable volume in each case (available water* gross value $/ML)	$	**41,076**	**65,037**	**65,037**	**65,037**	**65,037**	**65,037**	**65,037**
(e) Benefits of initial flood water (crop value/total flood volume)	$/ML	205.4	325.2	325.2	325.2	325.2	325.2	325.2
(f) Benefits of 200 ML of flood water harvested (benefit-cost)	$/ML	179.4	152.9	184.7	202.5	218.8	227.8	77.5
(g) Added Value % (above surface storage)		0 %	−15 %	3 %	13 %	22 %	27 %	−57 %

*multiplied by

recharging it underground. If the assumptions of this study are validated, aquifer storage using infiltration ponds would be financially viable.

17.5 Conclusion

Managed aquifer recharge can be a useful element of integrated water resource management. It can help to conserve surface water resources and improve groundwater quality (eg Adelaide case study), and minimize the evaporative loses and increase the volume of groundwater available for use (eg Namoi case study).

The economics depend on site-dependent factors. In general, recharge is least expensive where soils are permeable and aquifers are unconfined and fresh. Levelised costs may be approximately an order of magnitude less than the costs of recharge via wells. ASR is most cost efficient in aquifers that are transmissive and contain fresh or only mildly brackish ambient groundwater. It is attractive in urban areas where the value of recovered water is very high, it requires very small land area and if storing water in confined aquifers the groundwater resource is protected from overlying land uses.

MAR options have been shown to be economic in comparison with seawater desalination for urban substitutional supplies, and cheaper than use of mains water for public open space irrigation. Stormwater ASR has now been operational for 20 years in South Australia and the capacity is continually being expanded. For rural supplies the price of water is significantly lower than for urban supplies. And where infiltration rates are high and aquifers are unconfined, transmissive and contain fresh groundwater, it is possible for basin infiltration harvesting of supplementary entitlements during flood flows and their subsequent storage in aquifers to be a more efficient supply than harvesting in detention ponds alone that are exposed to significant evaporation losses. The Namoi desktop case study suggests that field validation of infiltration rates and maintenance requirements is warranted to determine the economics of MAR for flood water harvesting. Other such studies of recharge of flood waters such as Pavelic et al. (2012) in Thailand suggest that applications could potentially be very widespread.

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Part IV

Socioeconomics